Biodiversity Series

Literature Review on the Impacts of Dredged Sediment Disposal at Sea

OSPAR Commission

2008

The Convention for the Protection of the Marine Environment of the North-East Atlantic (the "OSPAR Convention") was opened for signature at the Ministerial Meeting of the former Oslo and Paris Commissions in Paris on 22 September 1992. The Convention entered into force on 25 March 1998. It has been ratified by Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, Netherlands, Norway, Portugal, Sweden, Switzerland and the United Kingdom and approved by the European Community and Spain.

La Convention pour la protection du milieu marin de l'Atlantique du Nord-Est, dite Convention OSPAR, a été ouverte à la signature à la réunion ministérielle des anciennes Commissions d'Oslo et de Paris, à Paris le 22 septembre 1992. La Convention est entrée en vigueur le 25 mars 1998. La Convention a été ratifiée par l'Allemagne, la Belgique, le Danemark, la Finlande, la France, l'Irlande, l'Islande, le Luxembourg, la Norvège, les Pays-Bas, le Portugal, le Royaume-Uni de Grande Bretagne et d'Irlande du Nord, la Suède et la Suisse et approuvée par la Communauté européenne et l'Espagne.

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ISBN 978-1-906840-01-3
Publication Number 362/2008

Literature review on the impacts of dredged sediment disposal at sea
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Executive Summary

The disposal of dredged sediments can affect the marine environment both through contaminants and physically, *e.g.* through smothering or habitat burial. This report provides an overview of information on the physical, chemical and ecological effects of the disposal of dredged material based on a study of available scientific literature. It summarizes the present knowledge, identifies scientific gaps and suggests directions for further research. It also provides summaries of studies on the effects of dumping of dredged material in the OSPAR Maritime Area (see Annexes).

Disposal of dredged sediments cause only minor chemical disturbances but scientific information is limited

The disposal of dredged material seems to cause no or only minimal chemical disturbances. All Contracting Parties have regulated the disposal of sediments at sea and only uncontaminated or slightly contaminated sediments that meet established environmental quality criteria (action levels) are disposed. However, it is noted that every Contracting Party defined its own action levels and that these levels can vary strongly. In addition, there are only action levels for a limited number of contaminants and e.g. none for “new” substances such brominated flame retardants.

Organic enrichment can affect benthos communities

Dredged sediment can contain a high content of organic matter. This organic matter can act as a food supply for benthic organisms and leads to communities dominated by opportunistic species such as annelids and nematodes.

Changes in the sediment structure can have adverse effects on benthic habitats

Habitat alterations were observed after the deposition of dredged sediment due to a change in sediment structure (i.e. grain size). The deposition of fine grained sediment on coarser grained natural sediment can lead to a reduced complexity and changes in community structures.

Enhanced sedimentation leads to burial and smothering of benthos communities

Direct burial under large quantities of dredged sediment often results in the immediate mortality of benthos. In cases where the amount of sediment is not too great, the effects are relatively small as many species are capable of migrating up through the deposited sediment. After deposition, the benthic community starts to recover or re-adjust. Recovery or re-adjustment rates of benthic communities following maintenance dredged material disposal ranges approximately between 1 month and 4 years. At disposal sites where dredged sediment is disposed more or less continuously, the benthic community does not fully recover.

Increased turbidity and suspended particulate matter concentrations might affect organisms but naturally occurring elevations are important too

Disposal of dredged sediments can lead locally to an increased turbidity. Increased turbidity might affect primary production, growth of macroalage and eel grass and visual predator fish species (e.g. herring and sprat) or fish eating bird species (e.g. tern species). However, naturally occurring turbidity elevations, induced by flood tides and weather activities, might even have a more significant effect than the periodic increased levels caused by disposal activities of dredged sediment. Increased suspended particulate matter concentrations may interfere with food intake of filter-feeding benthos and copepods, and functioning of gills of fish may be impaired due to clogging.

Knowledge gaps needs to be filled

Although a substantial amount of research has been conducted, some effects are not yet studied well. Without this information, the actual effects of the disposal of dredged sediments can not be assessed sufficiently. Further research should in particular focus on the: harmonization of national action levels for contaminants used by the individual Contracting Parties; the effects of changed redox potentials in former surface layers on benthic species (i.e. oxygen deficiency and sulphide production); the impact of increased turbidity due to disposal of dredged sediment on phytoplankton, eelgrass and visual predators; the effects of enhanced particulate suspended matter concentrations; criteria for selecting scientific reference sites

Impacts can be minimised

There are several possibilities to minimise the impacts of dredged sediment disposal. Dredged sediments should have similar sedimentary characteristics to those of the receiving site and be free from contaminants. The disposal method of dredged sediment should be adapted to natural processes and give motile macrofauna the opportunity to migrate horizontally to the new surface layers. The sediment should be disposed during the time of year when the impacts are minimal.

1. Introduction[1]

1.1 General information

Dredging harbours, docks and navigation channels, in order to deepen access conditions, is a long established human-induced disturbance in the marine environment. However, dredging is essential to maintain navigation in ports and harbours as well as for the development of port facilities. Most of the material dredged from within the OSPAR maritime area is, by its nature, either uncontaminated or only slightly contaminated by human activity (i.e. at, or close to, natural background levels) (OSPAR 1998). However, a smaller proportion of dredged material is contaminated to an extent that major environmental constraints need to be applied when depositing these sediments.

Annually, over 90 million tonnes (dry weight) within the OSPAR maritime area (OSPAR, 2005) and hundreds of millions of tonnes of sediments worldwide is disposed at sea and must be managed in an economically and environmentally sound manner (Bolam and Rees, 2003). It constitutes one of the most important problems in coastal zone management and in some coastal area represents the major anthropogenic disturbance to the benthos[2] (a.o. O'Conner, 1998; OSPAR 1998; Bolam and Rees, 2003). Because of concerns over potential environmental consequences of the disposal of maintenance dredging world-wide, its is becoming increasingly important to minimize impacts and, therefore, a good understanding of the impacts and most environmentally beneficial way to dispose of this material is now even more crucial (Bolam and Rees, 2003).

The effects of maintenance–dredged sediment disposal on the ecosystem (benthic community structure) has been well studied (a.o. reviews of effects by Engler *et al*., 1991; Essink, 1999; Fredette and French, 2004). Unfortunately most studies are not published in peer-reviewed and public domain journals. By reviewing both published and unpublished literature general conclusions about the impacts of maintenance–dredged sediment disposal can be drawn, and factors that control the impacts can be summarized.

1.3 International policy

In addition to the OSPAR Convention, two other conventions are of relevance for sediment management in Europe:

a. The **London Convention – LC** (1972) on the Prevention of Marine Pollution by Dumping of Wastes and Other Matter, which is signed by 75 countries around the world:

   Dredged material Assessment Framework – DMAF (1995)

b. The **Helsinki (HELCOM) Convention** (1992) on the Protection of the Marine Environment of the Baltic Sea Area:
   HELCOM RECOMMENDATION 13/1 (1992): Disposal of dredged spoil

The dredged material guidelines of all three conventions are harmonised to the greatest possible extent. Some countries have implemented these guidelines in special national guidelines like the *HABAK (Directive for the handling of dredged material in coastal waterways, 1999)* in Germany, the *Sea Disposal Framework of CEFAS* in England/Wales or the *Technical Rules for Excavation/Dredging and Management of Dredged Material (Ministry Order, n.º 141-June, 1995)* in Portugal.

1.2 OSPAR activities

The disposal of dredged sediments at sea affects the environment both through the contaminants it contains and also physically (OSPAR, 2006). For the environmental sound disposal of dredged material OSPAR Contracting Paris have agreed the 'OSPAR Guidelines for the management of Dredged Material (OSPAR ref. no. 1998-20)'. According to these Guidelines measures to keep the volume of dredged material to a minimum are regarded Best Environmental Practise (BEP) for minimising the effect on the environment.

In 2004 the 'Revised OSPAR Guidelines for the Management of Dredged Material' were adopted by the Contracting Parties (OSPAR, 2004). At present another revision is taking place and it is foreseen that newly revised Guidelines will be adopted by OSPAR 2009. These revised guidelines are designed to assist Contracting Parties in the management of dredged material in ways that will prevent and eliminate pollution

1 Information taken from SedNet (2003)
2 The benthos refers collectively to all aquatic organisms which live on, in, or near the bottom (substratum) of water bodies.

in accordance with Annex II to the 1992 OSPAR Convention, and protect marine species and habitats in the OSPAR maritime area in accordance with Annex V.

The issue of the impacts of disposal of dredged sediment to species and their habitats has been discussed at several meetings of OSPAR Committees and Working Groups and there are documents available in which this issue is addressed. There are, for example, guidelines available in the format for "Annual Reporting on 'Dumping' Operations at Sea" and an overall assessment of the disposal of waste (including dredged sediments) at sea in the period 1995 to 2002 was published in 2003.

In 2003 also a document called 'Contacting Parties' National Action Levels for Dredged Material' appeared. In this document the national action levels of the majority of the Contracting Parties were compiled. Large differences in the action levels of individual elements/compounds per Contracting Party were observed (for further details see paragraph 3.1.1). An updated version has been adopted by OSPAR 2008

Although many studies related to the disposal of dredged sediment are conducted, there was no overview on the (possible) impacts of disposal of dredged sediments in the OSPAR maritime region.

The publication of a JAMP assessment on the environmental impacts of dumping of waste is foreseen by OSPAR 2009.

1.4 Aims of this study

This study has the following aims:

1. To make an overview of the relevant current knowledge on the physical, chemical and ecological effects ofthe disposal of dredged sediments in the sea based on a literature review. The review is mainly focussed on direct effects on the disposal location and the surroundings, on long-term effects and on possible recovery of the environment after ending the disposal at a disposal site.
2. To summarize the measured and observed effects of the disposal of dredged sediment in the sea at four selected locations in more detail (selection based on information from literature review).
3. To summarize present knowledge, knowledge gaps and directions for future research.

2. Methodology of the state of the art research

This state of the art research is based on information from experts in the field of disposal of dredged sediment in the maritime environment. The following companies, universities and research institutes were contacted:

- CEFAS Burnham Laboratory (UK)
- Management Unit of the North Sea Mathematical Models (BE)
- Flemish Ministry of Mobility and Public Works (BE)
- Ministry of Environment. Danish Nature and Forest Agency (DK)
- Ministry of Environment. Danish Nature Environmental Protection Agency (DK)
- Ministère de l'Ecologie, du Développement et de l'Aménagement Durable (Fr)
- Bundesanstalt für Gewässerkunde (DE)
- Federal Agency for Nature Conservation (DE)
- Bundesamt für Seeschiffahrt und Hydrographie (DE)
- Ministry of Fisheries (IS)
- Department of Environment, Heritage and Local Government (IE)
- Marine Institute (IE)
- Ministry of Transport, Public Works and Water Management (NL)
- Directorate for Nature Management (NO)
- Institute of Nature Conservation (PT)
- Directorate-General for Biodiversity (ES)
- Ministry of Environment. Centre for Studies on Ports and Coasts (ES)
- Swedish Environmental Protection Agency (SE)

In addition to direct contacts with these experts, a literature search was performed on the internet and in the databases of university libraries. The libraries, databases and websites which were consulted are listed below. In addition the keywords used for the search are shown.

Libraries

- University of Utrecht
- National Institute for Coastal and Marine Management / *RIKZ*

Databases

- GeoRef
- Chemical Abstracts
- Omegasearch
- ScienceDirect

Websites

- www.cefas.co.uk
- www.rikz.nl
- www.scholar.google.com
- www.sednet.org
- www.uu.nl
- www.ospar.org
- www.nae.usace.army.mil/damos
- www.dvz.be
- www.zeeslib.nl

Keywords

- Dredged, sediment, material, spoil(s), dumping, disposal, sea, marine, coastal management, benthos, beneficial, use-schemes, turbidity, burial, smothering, physical, chemical, biological, characteristics, impact, effects.

The literature was screened for the following information:

- General characteristics: a.o. exact location of the disposal site (coordinates), surface are, water depth, turbidity, sedimentation rate and currents.
- Is the disposal site located in an OSPAR region? If so, what region?
- In which period took the disposal of the dredged sediment place?
- How much material is disposed (quantity)?
- What is the chemical and physical composition of the disposed sediment?
- What is the effect of the disposal on the turbidity at the disposal location and the surroundings?
- What is the effect on the grain size distribution at the disposal location?
- What is the effect of the dredged sediment disposal on the chemical composition of the sediment at the disposal site?
- What are the ecotoxicological effects of the disposal of dredged sediment at the disposal location (bioassays, effects on macrobenthos, megafauna, etc.)?
- Are there other effects –than stated above- observed (measured)?

This literature search does not pretend to be comprehensive. The most relevant literature has been collected and summarized. E.g. the individual references in already existing literature reviews are not mentioned separately and literature on impacts of dredged sediment disposal in tropical regions (a.o. Cruz-Motta and Collins, 2004) has not been taken into account.

3. Results

3.1 General

Over hundred publications and reports were found. Most studies are not published in peer-reviewed literature. All relevant studies – both conducted in and outside the OSPAR maritime area – are used to generate an overview on the impacts of the disposal of dredged sediments at sea. However, only the results of the studies conducted in the OSPAR maritime area are summarized in Annex 1 and tabulated in Annex 2-4.

The general characteristics of the reviewed maritime locations in the OSPAR regions, where dredged sediment has been disposed and monitored, are listed in Annex 2. General information regarding disposal activities and composition of the disposed dredged sediments in the OSPAR maritime areas is given in Annex 3. The measured and/or observed effects on sediment composition and fauna characteristics - in time and space – at disposal sites in the OSPAR regions, as found in the literature, are summarized in Annex 4. Ecotoxicological effects are not listed in Annex 3, since only three studies were found (BfG, 2001; Lauwaert *et al.*, 2006; Stronkhorst *et al*., 2003), conducted in the OSPAR maritime area. The results of the ecotoxicological tests of these studies are described in the text (see Annex 1).

The measured and observed effects of the disposal of dredged sediment in the sea at four selected locations are described in more detail in Annex 5.1 to 5.4. The selection is based on the quality and extensiveness of the research (spatial and temporal effects were studied). The selected sites are:

1. Netherlands Continental Shelf – Disposal sites Loswal North and Loswal Northwest (Annex 4.1);
2. United Kingdom Continental Shelf – Beneficial use sites Westwick Marina, Titchmarsh Marina and North Shotley (Annex 4.2);
3. United Kingdom Continental Shelf – Beneficial use site Titchmarsh Marina (Annex 4.3);
4. United States of America Continental Shelf – Charleston ocean dredged material disposal site (Annex 4.4).

Several effects of the disposal of dredged sediment at sea are distinguished in literature. The main effects are related to:

1. chemical disturbances
2. increased nutrient input
3. change in sediment structure
4. enhanced sedimentation (burial and smothering)
5. increased turbidity
6. enhanced suspended particulate matter

These effects are described in more detail below.

3.1.1 Effects due to chemical disturbances

Since all Contacting Parties forbid disposal of contaminated sediment at sea, only slightly to non-contaminated sediments are disposed. All contracting parties formulated national action levels for dredged sediment. Most Contracting Parties use a '3 category action level' approach which means that 2 concentration levels are provided (OSPAR, 2003). Concentrations of contaminants in dredged sediments below the lower limit represent those of little concern. Those falling between the lower limit and the upper limit may trigger further investigation of the dredged sediment. When concentrations of contaminants are higher than the upper value, disposal of dredged sediments at sea is often not permitted. The action levels are mostly based on baseline studies and toxicity studies.

Since dredged sediment may only be disposed at sea when the sediment composition meets the sediment quality criteria for disposal, the chemical impact of this sediment is considered to be zero or acceptably low. This is most likely the reason why in most studies the chemical impact of disposal of dredged sediment is not determined.

However, large differences in the action levels of individual contaminants per Contracting Party were observed (OSPAR, 2003). E.g. action level 2 for TBT[3] varies from 7 ppb in Belgium to 500 ppb in the United Kingdom. This means that dredged sediments with a TBT content of 500 are permitted to be dumped in the marine waters of the United Kingdom, but are not permitted to be dumped in the marine waters of Belgium. This means that a lot of dredged material in one Contracting Party can be disposed legally at sea - based on the established action levels - whereas the same lot of dredged material may not be disposed at sea in another Contracting Party. At the moment it is unclear whether or not these differences are acceptable - cause no impact to the marine environment.

The compilation of the national action levels for dredged material of the Contracting Parties also revealed that action levels are only established for a limited number of compounds and that this number can vary strongly between the Contracting Parties (OSPAR, 2003). In addition, no action levels exist for 'new' contaminants such as brominated flame retardants. It is therefore questionable if sufficient information is collected in current risk assessment studies to determine whether or not the disposal of dredged sediment will have a chemical impact or not.

In the following studies the chemical impacts of disposal of dredged material at sea are investigated:

[3] TBT = Tributyltin/ Disposal of sediment contaminated with TBT can cause imposex and intersex in gastropods potentially resulting in female sterilization. Imposex is the phenomenon of superimposition onto females of male sexual characteristics, such as a penis homologue and/or vas deferens.

1. BfG (2001) studied the effects of dredged sediment disposal on contaminant concentrations and toxicity in the Ems estuary in Germany. The content of heavy metals (As, Cd, Cu, Ni, Pb, Zn, Cr and Hg) and organic contaminants (PAHs, PCBs, TBT) in the disposed sediment is low. With the exception of Cu and HCB, the content of the heavy metals and organic contaminants were lower than the HABAK action level 1 (level at which the content of contaminants are of little concern – near background contaminant levels). The copper and HCB content were lower than action level 2 which means that there was no immediate cause to consider the restriction of the disposal activities. Ecotoxicological test showed that the dredged sediments were not toxic to slightly toxic. However, the dredged sediments were not or only slightly more toxic than the natural occurring sediments at the disposal site. BfG (2001) concluded that the disposal activities in the Ems estuary could continue.

2. Stronkhorst *et al*. (2003) studied the chemical effects of dredged sediment at the Netherlands Continental Shelf in detail. Concentrations of Cd, Hg, PCBs, PAHs and TBT in the fraction < 63 μm of the disposed dredged sediment were 2-3 times higher than at the reference site. However, levels of TBT were four times higher than the target value for imposex (Oehlmann *et al.*, 2000).

   In tissue (*pyloric caeca*) of resident starfish *Asterias rubens*, residual levels of Hg, Zn, PCBs and dioxin-like activity were never more than twice those at the reference site. In four different bioassays on starfish tissues, the sediments showed no acute toxic effects (Annex 1).

   Minor pathological effects were observed in resident dab *Limanda limanda*. Pathological effects were either within the range normally found in coastal waters of the Netherlands or differences were not significant and could not be related to causal constituents that are associated with the disposed sediment.

3. Lauwaert *et al.* (2004, 2006) determined the PCB, OCP, PAH, heavy metal and organotin content of the sediment at disposal sites and reference sites at the Belgian Continental Shelf. No difference between the disposal and reference sites was observed (Annex 3). The Cr, Ni, Cu and Pb content at one disposal site (ZO) is increasing significantly (fraction <2mm). However no relation could be established between this increase and dredging activities at the Belgian Continental Shelf. Two biochemical indicators (S.A. EROD and S.A. GSH-t) for pollution were determined by Lauwaert *et al.* (2006). No relationship between the biochemical indicators and disposal activities was observed.

4. Roberts and Forrest (1999) studied the effects of dredged sediment disposal on contaminant concentrations and toxicity to benthic macrofauna in the Tasman Bay (New Zealand). The dredged sediments were contaminated to varying degrees with some trace metals, organochlorine pesticides, PCBs and PAHs. Laboratory bioassays showed mildly elevated toxicity, and the macrofauna was dominated by small-bodied polychaetes. However, like in the studies of BfG (2001), Stronkhorst *et al*. (2003) and Lauwaert *et al.* (2004, 2006), there was very little indication of impact in the disposal area. The disposal area and control sites in Tasman Bay were all similar in terms of sediment contaminants, sediment toxicity and macrofauna. Roberts and Forrest (1999) describe the lack of discernable impact to the dynamic sedimentary environment in the disposal area, which disperses disposed sediments and mixes them with ambient sediment.

5. Based on 35 years of experience in New England (USA) with disposal of dredged sediment at sea, Fredette and French (2004) summarized the research efforts and resulting conclusions by the US Army Corps of Engineers. They conclude that impacts of organisms via the water column are generally minimal. Studies of mussel bioaccumulation have found that mussels usually show no significant bioaccumulation of contaminants. However, when significant bioaccumulation has been observed, contaminant levels of affected mussels returned to those at the reference locations shortly after cessation of disposal (see references in Fredette and French, 2004). Studies of reproductive tissue of mussels deployed at disposal sites also show little or no reproductive impairment compared to reference areas (see reference in Fredette and French, 2004).

Uncontaminated dredged sediments can also cause chemical changes. A covering layer of 1 m for example can change the redox conditions in the former surface layer considerably and anoxic conditions (oxygen deficiency and sulphide production) may develop shortly after the disposal (Essink, 1999). Bolam and Whomersley (2005) observed significant differences in the redox potential at 1 cm sediment depth at recharge location Titchmarsh Marina (Annex 4) one year after recharge. However, this change did not result in a measurable impact on the benthic fauna. Bolam and Whomersley (2005) did find a negative correlation between total individuals and 4cm redox potential (Annex 5.2). This means that (in theory) a change in redox potential (i.e. due to dredged sediment disposal) can affect benthic species.

BfG (2001) studied the effect of the disposal of dredged sediment on the oxygen content at the disposal site. They concluded that the disposal of dredged sediments did not adversely affect the oxygen content at the disposal site.

Based on the available studies it can be concluded that the disposal of dredged sediments causes no to minimal chemical disturbances. However, it is noted that in the majority of the studies the chemical disturbances are not investigated. Most likely, this effect is not well studied because dredged sediment may only be disposed at sea when the sediment composition meets the sediment quality criteria for disposal. If the chemical composition of dredged sediments comply with the sediment quality criteria, than the chemical impacts are considered to be absent or acceptable.

It is also noted that every Contracting Party defined its own action levels. These levels can vary considerably. It is unclear if the variance in action levels is acceptable in view of possible chemical disturbances due to the disposal of dredged sediment.

In addition it is noted that there are only action levels for a limited number of contaminants. This can be circumvented by conducting bioassays that are sensitive for a broad range of contaminants. In this framework the Netherlands developed the so-called Chemistry-Toxicity Test (CTT). Since 2004 the Netherlands used the CTT to assess whether the relocation of dredged material is acceptable. If it does not meet the quality criteria in the CTT, it is not allowed to be relocated in the marine environment (see DGE, 2007 for further details). Recently the CTT approach of the Netherlands was evaluated by Schipper and Klamer (2006). This included an assessment of the suitability of bioassays for disqualifying dredged sediment or performing an alert function. The performance characteristics of both the *Corophium volutator* and the Microtox Solid Phase test were not adjudged to be adequate for a disqualifying role in an assessment system such as the CTT. This also applies to any alert function in a monitoring system. The DR-CALUX test is however sufficiently robust to be used in a monitoring system for persistent, bioaccumulating and toxic substances.

As a consequence of the evaluation the CTT bioassays will no longer constitute part of the assessment system in the Netherlands.

3.1.2 Effects of nutrient input

Dredged sediment can contain a relatively high content of organic matter. This organic matter can act as a food supply for benthic organisms. In general, large increases in opportunistic species (a.o. worms) can occur in response to organic enrichment (Blanchard and Feder, 2003). This has been observed in Port Valdez (Alaska) following disposal of dredged sediment containing fish-wastes (Blanchard and Feder, 2003) and in the Anse a Beaufils, Baie des Chaleurs (Eastern Canada) following disposal of organic-rich dredged sediment (Harvey *et al.*, 1998). Zimmerman *et al.* (2003) reported an increase in the spionid polychaete P. *dayi,* classified as both a deposit feeder and a suspension feeder, depending on the availability of suspended and deposited particles. Zimmerman *et al.* (2003) conclude that the larger abundances of this organism in the impacted area might be the result of the increased organic matter content in the impacted area. Somerfield *et al*. (2006), conclude that organic enrichment leads to communities dominated by annelids and nematodes. This conclusion is based on reviewed studies which were conducted in various maritime areas.

It is concluded that dredged sediment with a relatively high content of organic matter can act as a food supply for benthic organisms which can result in an increase in opportunistic species (a.o. worms) at the disposal site. In this case, the disposal of dredged sediment results in a change in the benthic community structure.

3.1.3 Effects due to a change in sediment structure

Dredged sediment disposal can adversely affect the benthic community both by direct burial (see paragraph 3.1.3) and habitat alterations due to a change in sediment structure (Zimmerman *et al.*, 2003). Direct burial will often result in the immediate mortality of benthos. Habitat alteration can have long-term effects on the benthic community (Morton, 1977). Habitat alterations – mainly the reduction of habitat complexity – have been observed due to the deposition of fine-grained sediments on coarse grained natural sediments (BfG, 2001; Stronkhorst *et al.*, 2003; Zimmerman *et al.*, 2003; Van Dalfsen and Lewis, 2006).

BfG (2001), for example, observed a change in sediment composition after the disposal of dredged sediments. The grain size of the dredged sediments was in general lower (clay/silt) than the grain size of the natural occurring sediments (sand). This caused the macozoobenthic community to shift from a dominance of sand tolerant species to silt/clay tolerant species.

Stronkhorst *et al.* (2003) observed that the species richness and abundance of benthic invertebrates declined over an area extending about 1-2 km eastwards from the disposal site. This correlated with a shift in sediment texture form sand to silt.

Zimmerman *et al.* (2003) observed that after disposal, fine grained sediments dispersed to the west of the disposal site. In this area, the benthic community was strongly altered. The abundance of the cephalochordate *Branchiostoma* sp., for example, decreased significantly. *Branchiostoma* sp. is known to prefer sandy sediments and is seldom found in muddy areas (Boschung and Gunter, 1962; Cory and Pierce, 1967). The increase in silt and clay in the western area may explain the low abundance of *Branchiostoma* sp. And consequently the low abundance of 'other taxa' in this area.

Two years and 3 months after disposal of dredged sediment in the Verdiepte Loswal, the macrofauna community at the disposal site stills differs from the reference location (Van Dalfsen and Lewis, 2006). The reference location is characterized by lower number of species and different species compared to the species at the former disposal site. In addition, the species are also more evenly distributed at the reference location. Crustaceans are an important group at the reference location. Van Dalfsen and Lewis (2006) conclude that the differences in the macrofauna community are related to the differences in the sediment composition (a.o grain size). According to the researchers, complete recovery will only take place if the sand content of the sediment at the former disposal site increases.

It is concluded that dredged sediment disposal can adversely affect the benthic community if the sediment structure of the dredged sediments differs too much from the sediment structure of the natural occurring sediments at the disposal site. Negative effects due to a change in sediment structure can be minimized by selecting receiving sites that have similar sedimentary characteristics as the dredged sediments to be disposed.

3.1.4 Effects due to enhanced sedimentation

Enhanced sedimentation is the most cited impact of dredged sediment disposal. In all reviewed (accessible) studies, it is concluded that excessive deposition of dredged sediment leads to burial, smothering, or crushing of the benthos. Most benthic organisms live in the top 10 cm of the seabed and must maintain some connection to the sediment-water interface for ventilation and feeding (Miller *et al.*, 2002). This connection is disturbed by excessive sediment deposition.

Colonization of impacted environments by the assemblages following dredged sediment disposal may occur via one or more of three main mechanisms (Bolam and Rees, 2003):

1. vertical migration of the buried individuals through the dredged sediment,
2. horizontal immigration of postlarval individuals from the surrounding communities and,
3. larval recruitment from the water column.

In cases where the amount of sediment disposed is not too great, the effects are relatively small as many of the species are capable of migrating up through the deposited sediments (a.o. Bijkerk, 1988; Essink, 1999; Schratzberger *et al.*, 2006; Wilber *et al.*, 2007). Often however, the amount deposited is too great to allow species to survive burial and recovery occurs via recolonization of and/or immigration to the new sediment surface (a.o. Stronkhorst, 2003; Bolam *et al.*, 2006). The long-term effects in such cases may be more severe since recovery of benthic communities, a major food source for many other animals (e.g. fish) will be more prolonged.

Negative effects on benthos at disposal sites can be reduced by disposing in such a way that the layer of deposited sediment does not exceed 20 to 30 cm (Essink, 1999). Wilber *et al.* (2007) proposed the disposed sediment layer not to exceed 15 cm (in thickness). Most likely the suggested thickness varies because the thickness of the disposed sediment layer at which benthic species survive is species dependent and depends on the characteristics of the sediment. Bijkerk *et al.* (1988) determined this thickness for a variety of individual macrobenthic species.

Negative effects due to burial and smothering of the benthos can also be minimized by selecting bottoms which are poor in benthic life (Essink, 1999).

3.1.5 Effects due to increased turbidity[4]

Disposal of dredged sediments will cause local and temporal (re)suspension of sediments, causing increased turbidity. High turbidity results in low levels of transmitted light and can therefore negatively affect functioning of light-dependent organisms such as phytoplankton, eelgrass and visual predators, e.g. fish and fish-eating birds (Essink, 1999). Increased turbidity can be both caused by natural processes, such as storm events and tides, and human activities, e.g. the disposal of dredged sediment at sea.

Phytoplankton production is directly dependent on light penetration into the water column. Increased water turbidity results in a decrease in light penetration which is likely to affect phytoplankton adversely (Essink, 1999). However this effect will be rather local and restricted in time, and therefore have little effect on total primary production of an estuary or of a tidal basin in which disposal took place.

The occurrence and growth of eelgrass is highly dependent on the transparency of the water. Decrease in light penetration into the water due to sediment disposal may therefore impair conditions for growth of eelgrass and other macrophyte species (Essink, 1999).

For visual predators light plays an important role in finding, recognizing and capturing prey. An increase in water turbidity may negatively influence the performance of a visual predator, not only by the decrease in light intensity but also by changes in the spectral composition and polarisation pattern of the light (Essink, 1999). Visual predators that are known to avoid turbid waters are herring and sprat. Fish-eating birds use their eyes in chasing and capturing prey under water. For these species it is not known whether turbidity of the water affects the foraging success. However the sandwich tern needs clear water in order to locate its prey, therefore negative effects due to an increase in turbidity are to be expected.

Lauwaert *et al.* (2004) determined the influence of the disposal of dredged sediment at the Belgium Continental Shelf on the turbidity and concluded that the disposal results in a local enhancement of the turbidity. However, this effect is comparable low compared with the natural turbidity due to tides and weather activities. This agrees with the conclusions of Hydrographic Surveys Ltd (2003). They conclude that naturally occurring turbidity elevations, induced by flood tide, have a more significant and long term effect than the periodic increased levels caused by disposal activities of dredged sediment.

The disposal of dredged sediment in the Ems estuary – Germany - results locally in a slight to strong temporary increase in turbidity (BfG, 2001). BfG (2001) concludes that as a result of the high energetic environment and the naturally occurring high suspended particulate matter (SPM) concentrations, the disposal of dredged material will not lead to permanent higher SPM concentration in this area.

Both Lauwaert *et al.* (2004) and BfG (2001) determined the fish quantities at the disposal sites. In both studies is concluded that the disposal activities had no adverse effects on the fish quantities. This infers indirectly that fish were not adversely affected y the increased turbidity at the disposal sites in Belgium and Germany.

Turbidity studies conducted by the U.S. Army Waterways Experiment Station using adult marine, estuarine and freshwater organisms have shown lethal concentrations of suspended dredged material to be an order of magnitude or more higher than maximum water-column concentrations observed in the field during dredging operations (Engler *et al.*, 1991). Field observation following disposal operations have shown turbidity or suspended particulate levels to be less than 1 g/l that persisted for exposure times of only hours (Engler *et al.*, 1991). Based on these and other observations, Engler *et al*. (1991) concluded that: (1) the physical effect of turbidity from dredged sediment disposal in open water would be of minimal impact and (2) the primary impact of turbidity is aesthetic and must be treated as such. A major exception of this would be the sensitive coral reefs of tropical waters (Engler *et al.*, 1991). However, these don’t occur in the waters of the OSPAR maritime area.

It is concluded that disposal of dredged sediments will cause local and temporal (re)suspension of sediments, causing increased turbidity. However, it is also concluded that naturally occurring turbidity elevations, induced by flood tides and weather activities, have a more significant effect than the periodic increased levels caused by disposal activities of dredged sediment. This means that the impact of dredged sediment disposal of light-dependent organisms due to increased turbidity will most likely not have a greater impact than naturally occurring turbidity elevations, induced by flood tides and weather activities. By selecting time windows during which dredged sediment may be disposed of, possible adverse effects of

[4] Turbidity is the measure of the light scattering properties of water and depends on the amount, size and composition of the suspended matter such as clay, silt, colloidal particles, plankton and other microscopic organisms. It is measured in nephelometric turbidity units (NTU). Turbidity is easy to measure and is sometimes used as a surrogate for suspended particulate matter, but this is not straightforward.

increased turbidity can be minimized (Essink, 1999). See Annex 6 for examples of the possible adverse effects of increased turbidity with respect to the time of disposal.

3.1.6 Enhanced suspended particulate matter[5]

During disposal of dredged sediment at sea large amounts of sediment are brought into suspension (Essink, 1999). Increased suspended particulate matter (SPM) concentrations may interfere with food intake of filter-feeding benthos (bivalves) and copepods, and functioning of gills of fish may be impaired due to clogging (Essink, 1999). According to Widdows *et al.* (1979) growth of filter-feeding bivalves may be impaired at SPM concentrations > 250 mg/l.

Three studies were found in which this effect was studied. The results of the studies contradict. De Jonge and De Jong (2002) examined the possible relationship between SPM in the Wadden Sea (The Netherlands) and dredging operations. The major short-term variations in annual mean SPM in part of the Wadden Sea appears to be a non-linear, exponential, function of (a.o.) dredged sediment disposal. Without any disposal of dredged sediment, De Jonge and De Jong (2002) expect SPM concentrations in the tidal inlet channels of the Wadden Sea to be <15 g/m^3 (comparable to the 1950s). The overall mean annual SPM concentration for the investigation period – with disposal of dredged sediment - reached 35 to 42 g/m^3 (location dependent).

The disposal of dredged sediment in the Ems estuary – Germany - results locally in a slight to strong temporary increase in SPM concentrations (BfG, 2001). Increased SPM concentrations were measured to 2300 m from the disposal site. As a result of the high energetic environment and the naturally occurring high SPM concentrations, BfG (2001) argues that the disposal of dredged material will not lead to permanent higher SPM concentrations in this area. BfG (2001) determined the fish quantities at the disposal sites. In this study it was concluded that the disposal activities had no adverse effects on the fish quantities. This infers indirectly that fish quantities were not adversely affected by the increased turbidity at the disposal sites in Germany.

Dronkers (2007) concluded in a study in which the effects of the extension of the Tweede Maasvlakte (The Netherlands) are studied that disposal of dredged sediment does not seem to have a measurable influence on SPM concentrations. This is the opposite of what BfG (2001) and De Jonge and De Jong (2002) conclude.

Based on the limited information no conclusion can be drawn - besides the comment that this effect may occur and might be *site-specific*. In addition, the possible effects of increased SPM concentrations on vulnerable species such as bivalves and copepods have not been studied.

Bolam *et al.* (2006-2) used a number of numerical techniques to assess impacts at 18 different disposal sites. Their analyses revealed that ecological effects associated with dredged sediment disposal were *disposal-site specific*. Therefore they conclude that any assessment of the consequences of dredged sediment disposal at sea must take account of site-specific variation in prevailing hydrographic regimes and in ecological status, along with information on the disposal activity itself (mode, timing, quantity, frequency and type of material).

3.1.7 Other effects

Other factors that can affect the impact of dredged sediment disposal, besides the presence of toxic compounds in the dredged sediments, the sediment structure of the disposed dredged sediment, the thickness of the disposed sediment layer, increased turbidity, enhanced suspended particulate matter concentrations and enhanced nutrient input with the disposal of the dredged sediment are:

- Depth of water at the receiving site (Toumazis, 1995)
- Period of burial (Maurer *et al.*, 181a,b, 1986; Chandrasekara and Frid, 1998; Schratzberger *et al.*, 2000)
- Ambient temperature (Maurer et al., 181a,b, 1986; Chandrasekara and Frid, 1998).

These factors were not studied in field surveys, but several laboratory experiments were conducted to study the effects of these factors on benthic species.

Toumazis (1995) postulates that disposal of dredged sediments at depths greater than the reach of light will significantly harm marine organisms. Organisms in such depths are not accustomed to changes of their environment and their ability to survive and recolonise the new material is doubtful. However, the literature review did not yield any information to validate this thesis.

[5] Suspended particulate matter refers to the mass of suspended solids in water and is measured as mg/l. Turbidity is sometimes used as a surrogate for suspended particulate matter, but this is not straightforward.

Schratzberger *et al.* (2000) studied the effects of simulated deposition of dredged material on structure of nematode assemblages. They found that the response of nematode assemblages was mainly determined by the deposition frequency rather than the type of sediment or the degree of contamination. The deposition of sediment in one large dose at the beginning of the experiment caused more severe changes in assemblage structure than the same quantity deposited in several smaller doses. Higher migration and survival rates were reported in the high-frequency treatment than in the single dose.

Maurer *et al.* (1981a, b, 1986) conducted a laboratory assessment in which they investigated the ability of 3 estuarine mollusc to migrate vertically in natural and exotic sediments. They found that mortalities were higher with increased depth (read: thickness of the sediment layer) and with increased duration of burial. Mortalities were also higher at summer temperatures than winter, possibly due to oxygen stress in the sediment during summer. Nevertheless, the vertical migration of the fauna was higher at the summer temperature. Chandrasekara and Frid (1998) conducted a series of laboratory experiments to investigate the effects of burial on two epibenthic gastropod species under various sediment temperature regimes. The proportion of one gastropod species (*H. ulvae*) surviving burial in natural sediment to 5 cm depth decreased with increasing duration of burial and increasing temperature. Burial to 5 cm was fatal to the other gastropod species (*L. littorea*) within 24h at all the temperatures examined.

3.2 Spatial and temporal extent of impacts

Although the amount of 'accessible' research is limited, impacts to the benthic community at disposal sites typically are near-field and short-term (a.o. Leuchs and Nehring, 1996; Bolam and Rees, 2003; Stronkhorst *et al.*, 2003; CEFAS, 2005; Bolam *et al.*, 2006).

3.2.1 Spatial extent of the impact

At disposal site Loswal Northwest, located at the Netherlands Continental Shelf, Stronkhorst *et al.* (2003) determined the impacts of sediment disposal from the disposal site till 8 km eastwards. During the time of disposal, the species richness and abundance of benthic invertebrates declined over an area extending about 1-2 km eastwards (Annex 4).

Leuchs and Nehring (1996) determined the spatial impact of the disposal of dredged sediment in the Elbe Estuary, Germany. They showed that the disposal had an impact on macrozoobenthos in an area extending about 1000 m upstream and downstream of the official disposal site (Annex 4).

Zimmerman *et al.* (2003) studied the spatial and temporal effects of disposal of dredged sediment in the Charleston Ocean (USA). They concluded that the disposal of fine-grained inner harbour sediments into the Charleston ocean disposal zone (USA) had resulted in physical and (adverse) biological effects in areas surrounding the disposal zone. The effects were measurable to approximately 1.5 mile from the actual disposal zone. The results of this study are described in more detail in Annex 5.4.

Fredette and French (2004) summarized the research efforts and resulting conclusions by the US Army Corps of Engineers, based on 35 years of experience in New England (USA) with disposal of dredged sediment at sea. They conclude that the only discernible adverse impacts have been near-field and short-term. Physical monitoring has revealed that disposed sediments are quickly transported to bottom, and short-term losses of sediment due to dispersion are only 1-5% of total sediment deposited. Impacts to the benthic community have been carefully studied employing a variety of techniques. Direct effects of disposal have been detected only within a few hundred meters of the disposal point (Fredette and French, 2004). Further from the disposal point, where only thin (<50cm) layers of sediment are deposited, benthic organisms can burrow through the deposited sediment layer.

3.2.2 Temporal extent of the impact

In many studies the impacts of disposal are monitored in time (a.o. Harvey *et al.* 1998; Blanchard and Feder, 2003; Stronkhorst *et al.*, 2003; Bolam *et al.*, 2006 and review of recovery rates in Bolam and Rees, 2003). Often these studies are conducted to determine the time needed for the fauna to recover. De Grave and Whitaker (1999) suggest that recovery is not a suitable term to apply when assessing re-colonization after a disturbance since recovery implies return to faunal compositions and associated ecological pathways developed over many years (Blanchard and Feder, 2003). They suggest that 're-adjustment' rather than recovery is the appropriate terminology. In this paragraph both terms are used interchangeable.

McCall (1977) and Rhoads *et al.* (1978) describe general stages of benthic re-adjustment following major disturbance, under which dredged sediment disposal. Stage I: the benthos is barren of invertebrates, Stage II: recruitment of opportunistic taxa and faunal abundance increases exponentially, and Stage III: opportunistic species are replaced by larger, slower-growing 'equilibrium species'.

Bolam and Rees (2003) reviewed the recovery rates of benthic invertebrates following one-time placement of dredged sediment[6]. Recovery of benthic communities following maintenance dredged material disposal in coastal, polyhaline environments generally ranges from 1 to 4 years (Bolam and Rees, 2003). Although the reviewed ecosystems experience more or less constant salinities, physical stresses (mainly due to wave action and tidal currents) vary greatly. Differences in physical stresses appear to be correlated with recovery times (Bolam and Rees, 2003). While deeper sites tend to take at least 2 years to recover, shallower or more physical stressed sites take noticeably less time (less than 1 year in many cases) to return to the initial community structure. In euhaline coastal environments recovery times are also correlated with the degree of natural physical disturbance or depth (Bolam and Rees, 2003). While recovery rates in shallower sites are generally between 1 to 9 months, deeper sites may take up to 2 years to recover.

Bolam and Rees (2003) developed a conceptual model of changes in invertebrate assemblages following dredged sediment disposal in the marine environment. This model is visualized in Figure 3.1.

The model summarizes the natural macrofaunal assemblages found at different levels of environmental stress (e.g. wave action, salinity changes) together with the recovery mechanisms for each habitat type following dredged sediment disposal (Bolam and Rees, 2003).

The recovery rate of the species richness, (relative) abundance and diversity in the OSPAR maritime area varies from 3 months (Bolam *et al.*, 2006) to more than 2 years (Stronkhorst *et al.* 2003; Bolam and Whomersley, 2005; Van Dalfsen and Lewis, 2006)(Annex 4). At some disposal locations in the OSPAR maritime area, the benthic community never recovers, because sediment is disposed there more or less continuously. Repeated disturbances, such as described by Leuchs and Nehring (1996)(Annex 3), to a benthic system results in a succession that never proceeds beyond the initial re-adjustment phase (Stage I).

Community structures, however, often fail to converge with reference sites within the monitoring period (no recovery). After 4 years the community structure at the disposal sites studied by Bolam *et al.* (2006) in the United Kingdom, was still not recovered (Annex 4). This has also been observed by Johnson and Nelson (1985), Bolam and Whomersley (2005) and Wilber *et al.* (2007). In these studies species richness, (relative) abundance, biomass, diversity, etc. recovered, but community structure did not. Bolam *et al.* (2006) argue that this failing of convergence (to reference conditions) in community structure is caused by large spatial differences in reference locations in highly dynamic areas rather than a permanent adverse impact caused by the disposal of dredged sediment. Recovery time estimates, therefore, depend not only on biological responses, but the analytical approach used to evaluate and report the data. Selection of suitable reference sites is essential to be able to determine the recovery rates of the benthic community after disposal of dredged sediment.

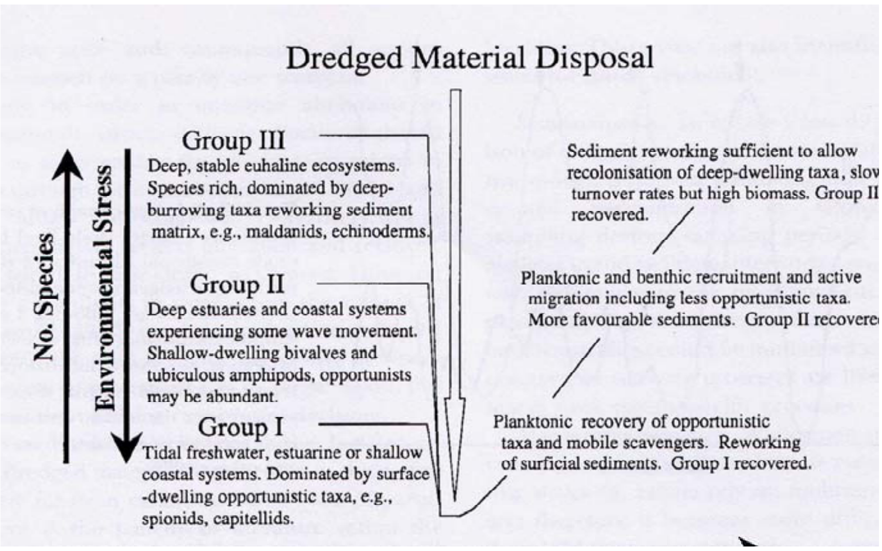

Figure 3.1 Conceptual model of changes in invertebrate assemblages following dredged sediment disposal in the marine environment (Source: Bolam and Rees, 2003).

6 The data are predominantly from the United States, where most research has been conducted to investigate benthic recovery processes following dredged sediment disposal and how they compare between different receiving systems (Bolam and Rees, 2003).

4. Conclusions

4.1 Impacts due to disposal of dredged sediment at sea

The following impacts can occur in the OSPAR maritime area:

- Since all Contracting Parties forbid disposal of contaminated sediment at sea, only sediments with a sediment composition that meets the established environmental quality criteria (action levels) are disposed. This is most likely the reason for the limited information on the chemical impacts of disposal of dredged sediment in the OSPAR maritime area. Studies on the chemical effects of the disposal of dredged sediment are most likely considered as redundant.

  Based on the available studies can be concluded that the disposal of dredged sediments causes no to minimal chemical disturbances. However, it is noted that every Contracting Party defined its own action levels. These levels can vary strongly. It is unclear if the variance in action levels is acceptable in view of possible chemical disturbances due to the disposal of dredged sediment. In addition, there are only action levels for a limited number of contaminants. To circumvent this problem, bioassays (CTT approach) were used in the Netherlands to detect a broad range of toxic compounds. However, evaluation of the CTT approach in 2006 revealed that CTT bioassays are not suitable for disqualifying dredged sediments. Therefore they do not longer constitute part of the assessment system in the Netherlands.

- Dredged sediment can contain a relatively high content of organic matter. This organic matter can act as a food supply for benthic organisms. Increases in opportunistic species (a.o. worms) have been observed in response to organic enrichment due to the disposal of organic-rich dredged sediment in Alaska and Canada.

- Habitat alterations are observed after the deposition of dredged sediment due to a change in sediment structure (a.o. grain size). Complexity is found to be reduced and community structures have been changed due to the deposition of fine grained sediment on coarser grained natural sediment.

- Enhanced sedimentation due to the disposal of dredged sediment causes burial and smothering of the benthic community. In all reviewed studies is concluded that benthic communities are adversely affected by the disposal of dredged sediments due to burial and smothering. Direct burial under large quantities of dredged sediment often results in the immediate mortality of benthos. In cases where the amount of sediment is not too great, the effects are relatively small as many species are capable of migrating up through the deposited sediment. After deposition, the benthic community starts to recover or re-adjust. Recovery or re-adjustment rates of benthic communities following maintenance dredged material disposal ranges approximately between 1 month and 4 years. At disposal sites where dredged sediment is disposed more or less continuously, the benthic community does not fully recover.

- Disposal of dredged sediments will cause local and temporal (re)suspension of sediments, causing increased turbidity. Several researchers determined the influence of the disposal of dredged sediment on the turbidity. It was concluded that naturally occurring turbidity elevations, induced by flood tides and weather activities, have a more significant effect than the periodic increased levels caused by disposal activities of dredged sediment. This means that the impact of dredged sediment disposal on light-dependent organisms due to increased turbidity will most likely not have a greater impact than naturally occurring turbidity elevations, induced by flood tides and weather activities.

- During disposal of dredged sediments at sea large amounts of sediments are brought into suspension and may cause increased suspended particulate matter (SPM) concentrations. Three studies conclude the opposite with regard to the possible impact on SPM concentrations: significant increase of SPM concentrations versus no measurable influence on SPM concentrations. Most likely this effect is site specific, weather depended and influenced by the disposal method.

- In several studies the spatial extent of the effects of the disposal of the dredged sediment was studied. These studies showed that the disposal of dredged sediment can have an impact on (macrozoo) benthos in an area extending about 2 km from the official disposal site. This is most likely (a.o.) site specific, weather dependent and influenced by the disposal method.

The following additional effects have been observed in laboratory experiments, but have not been verified in field experiments:

- Deposition of sediment in one large dose causes more severe changes in assemblage structure than the same quantity deposited in several smaller doses. Higher migration and survival rates were reported in high-frequency treatment than in a single dose.
- Mortalities of molluscs were higher with increased depth (read: thickness of the sediment layer) and with increased duration of burial. Mortalities were also higher at summer temperatures than winter, possibly due to oxygen stress in the sediment during summer. However the vertical migration of the molluscs was higher at the summer temperature.

4.2 Recommendations

Based on the results of this study it can be concluded that physical, biological and chemical impacts do occur, but are localised within or close to the boundaries of the disposal site. The impacts are inherent to the disposal activities of dredged sediment. The Contracting Parties should determine whether or not these impacts are acceptable. This decision can be made best if the following information becomes available:

1. Lacking information on the impacts of disposal of dredged sediment to fill the knowledge gaps.
2. Comparison with other impacts, both natural and man-induced, in the marine environment.
3. Management options to minimise the impact of the disposal of dredged sediments at sea.

4.2.1 Knowledge gaps

Although a substantial amount of research has been conducted, some effects are not yet studied well. Without this information, the true extent of the impact of the disposal of dredged sediments can not be determined well. Knowledge on the following subjects is desirable:

- The action levels of contaminants, used by the individual Contracting Parties, can vary strongly. This means that a lot of dredged material in one Contracting Party can be disposed legally at sea - based on the established action levels - whereas the same lot of dredged material may not be disposed at sea in another Contracting Party. At the moment it is unclear whether or not these differences pose a threat to the marine environment.
- In the literature it is postulated that a covering layer of dredged sediments can change the redox conditions in the former surface layer considerably and anoxic conditions (oxygen deficiency and sulphide production) may develop shortly after the disposal. In the literature it is postulated that change in redox potential can affect benthic species. However, the impact of these effects is not well studied and quantified.
- Disposal of dredged sediments will cause local and temporal (re)suspension of sediments, causing increased turbidity. In turn, increased turbidity may impair functioning of organisms such as phytoplankton, eelgrass and visual predators (e.g. fish and fish-eating birds). Although researchers conclude that naturally occurring turbidity elevations induced by flood tides and weather conditions have a more significant effect, turbidity due to disposal of dredged sediment might have an adverse effect on phytoplankton, eelgrass and visual predators under certain conditions (e.g. stable weather conditions in summer). No information has been found on the possible impact of increased turbidity due to disposal of dredged sediment on phytoplankton, eelgrass and visual predators.
- Studies on the effects of the disposal of dredged sediment on SPM concentrations are limited and the results of the available studies are contradicting. Increased SPM concentrations may interfere with food intake of filter-feeding benthos (bivalves) and copepods, and functioning of gills of fish may be impaired due to clogging. No studies on this effect have been found.
- An important issue that would be good to resolve in sediment management, and one that is addressed differently by the various Contracting Parties (EU Member States) is how one identifies reference sites. Reference sites can be used to make comparisons among biological, chemical or physical sediment data that might be collected from an area under study. Lack of appropriate criteria for selecting reference areas may result in an inappropriate location being selected, and inappropriate sediment management actions being undertaken. Identification of the reference site may depend on the remediation goals and options, historical and existing conditions at the site, as well as critical physical, chemical and biological parameters that are being evaluated.

4.2.2 Comparison with other impacts

The disposal of dredged sediment at sea is not the only disturbance of the marine environment. Other disturbances, both natural and human, are ship traffic, fishing, tides or storm events. The impacts of these disturbances can be far greater than that of dredged sediment disposal. By comparing the impacts of a variety of human activities and natural events at sea, the impacts of dredged sediment disposal can be placed in perspective.

Dredged sediments can also be used to derive environmental or other benefits. One current practice is the use of dredged sediments for habitat creation and/or restoration (i.e. beneficial use). Dredged sediments have already been shown to successfully enhance or create new mudflats. However, the physical, chemical and biological impact of dredged sediments when used for habitat enhancement (i.e. beneficial use) is still under investigation. Some researchers advocate that (future) studies aiming to investigate the community development of beneficial use schemes should adopt large numbers of stations sampled singly, rather than smaller numbers of stations each samples several times (more spatial data than temporal data).

4.2.3 Minimise impacts

There are several possibilities to minimise the impacts of dredged sediment disposal. To minimise impacts, management options for the location and time of disposal can be formulated. Disposal of dredged sediment at sea will have minimal impact if:

1. Dredged sediment had similar sedimentary characteristics to those of the receiving site
2. Dredged material is free from contaminants;
3. The disposal of dredged sediment is adjusted to the natural processes. By quantifying natural sedimentation rates and the susceptibility of macrofauna to these natural processes, environmental impacts of dredged sediment disposal can be better predicted and disposal schemes can be tuned to these natural processes.
4. The disposal method systematically distributes a number of shallow layers of sediments over the disposal site and thus motile macrofauna had the opportunity to migrate upwards between passes of the barge.
5. The sediment is disposed during the time of year – so-called time windows – that the impacts are minimal.

Essink (1999) formulated various management options for disposal and linked this to the possible effects and causes of the effects. The management options of Essink (1999) are listed in Annex 6.

However the question of where and when to dump dredged sediments is still mainly determined by economical considerations: 1) the costs for shipping the dredged sediments to the disposal site and 2) the chance of recirculation of disposed sediments to the original dredge site.

Because impacts are site specific, potential environmental effects of each disposal project must be evaluated on a case-by-case basis.

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Annex 1 Summaries of the studies on the effects of disposal of dredged sediments in the coastal environment, conducted in the OSPAR maritime area.

Rees *et al*. (1992) studied the sediments and benthic fauna at dredged sediment disposal sites in inner Liverpool bay (United Kingdom). Contrary to the expectation, there was no evidence of a widespread area of faunal impoverishment in the immediate vicinity of the disposal site, even at high disposal rates (several million tonnes per year) in the early 1970s. Newly deposited sediment is rapidly recolonized by the larger 'opportunistic' species. As a result, the (longer-term) main effect of disposal appears to be one of enhancement in numbers. This may occur both as a result of larval recruitment, or redistribution of adults. It is also probable that, once established, populations on the periphery of the disposal site can survive repeated additions of migrating dredged materials.

Kleef *et al.* (1992) studied the effects of the disposal of dredged sediment on the benthos in the Old Westereems (The Netherlands) between 1989 and 1990. The sediment characteristics of the disposal site and reference area were different (Annex 2 and 4). The percentage of silt (fraction < 16 μm), clay (fraction <2μm), calcium carbonate and organic matter were higher at the disposal site (just not significantly). The most important ecological effects were a significant reduction in the number (approximately 50%) and density of species (Annex 4). For *Nephtys hombergii* this decrease was correlated with the thickness of the disposed layer. One year after disposal, the bottom fauna was recovering. However, before recovery was complete, the next layer of dredged sediment was deposited.

Leuchs and Nehring (1996) described that, since the opening of the Kiel Canal in 1895, mud has been dredged routinely from the Brunsbüttel locks and the inland port and disposed almost daily in more or less the same area of the Elbe estuary (Germany). The disposal had a proven impact on macrozoobenthos in an area extending about 1000 m upstream and downstream of the official disposal site. An investigation of benthos showed an impoverished macrobenthic fauna throughout the 1000 m disposal area, in comparison with the downstream reference stations (Annex 4). Because of the frequent disposal, Leuchs and Nehring (1996) expect the macrobenthic fauna to remain in a stage of continual recolonization with few, rapidly growing species having high production rates. Permanent colonization with long-lived species in the disposal area is not to be expected.

Essink (1999) conducted a literature review, summarized the ecological effects of dredged sediment disposal and formulated management options based on the literature results. Essink (1999) concluded that disposal of dredged sediments in estuarine and coastal waters may lead to increased turbidity and enhanced sediment deposition at disposal sites. This mainly affects primary production by phytoplankton, performance of visual predators (e.g. fish, birds) and growth and survival of benthic organisms. Increased turbidity may affect dab[7] as well as prey location by sandwich terns[8]. Enhanced suspended particulate matter (SPM) concentrations are unfavourable for young herring and smelt. Growth of filter-feeding bivalves may be impaired, especially at SPM-concentrations > 250 mg/l. Estuarine nematodes can survive burial by 10 cm of disposed dredged sediment provided that its physical characteristics are similar to those of the original sediment. Sessile benthos organisms such as mussels and oysters can cope with sediment deposition of only 1-2 cm. Other macrozoobenthos can survive sediment deposition of 20-30 cm. Recovery of benthos at a disposal site will occur if the interval between successive disposals is sufficiently long.

A German consortium of research groups studied the physical, chemical and biological effects of the disposal of dredged sediments in the Ems estuary in the period 1999 to 2000 (BfG, 2001). Approximately 14 million m^3 of dredged sediments was deposited. The disposal of dredged sediment in the Ems estuary results locally in a slight to strong temporary increase in turbidity and SPM concentrations. As a result of the high energetic environment and the naturally occurring high SPM concentrations, BfG (2001) argues that the disposal of dredged material will not lead to permanent higher SPM concentration in this area. The content of heavy metals (As, Cd, Cu, Ni, Pb, Zn, Cr and Hg) and organic contaminants (PAHs, PCBs, TBT) in the disposed sediment is low. With the exception of Cu and HCB, the content of the heavy metals and organic contaminants were lower than the HABAK action level 1 (level at which the content of contaminants are of little concern – near background contaminant levels). The copper and HCB content were lower than action level 2 which means that there was no immediate cause to consider to restrict the disposal activities. Ecotoxicological test showed that the dredged sediments were not toxic to slightly toxic. However, the dredged sediments was not or only slightly more toxic than the natural occurring sediments at the disposal

[7] Dab = flat fish (*Limanda limanda*)
[8] Sandwich tern = bird belonging to the visual predators.

site. Therefore it was concluded that the disposal activities could continue. The disposal of dredged sediments did not adversely affect the oxygen and nutrient content at the disposal site.

At several disposal sites in the Ems estuary the structure of the macrozoobenthic communities changed due to the disposal of dredged sediments. The changes were caused by the change in sediment composition. The grain size of the dredged sediments was in general lower (clay/silt) than the grain size of the natural occurring sediments (sand). Therefore, the macozoobenthic community shifted from a dominance of sand tolerant species to silt/clay tolerant species. The disposal activities had no negative impact on the fish and shrimp quantities. BfG (2001) concluded that the disposal of dredged sediments in the Ems estuary has hardly any ecological effects. Based on these insights the disposal activities can continue under the employed conditions.

An Irish consortium of research groups studied the physical, chemical and biological effects of the disposal of dredged sediments in Galway Bay (Galway Bay, 2003). Several weeks after disposal of the dredged sediments the Zn, Pb, Ni, Mn, Fe, Cu and organic carbon content of the sediment at the disposal site was increased significantly (Annex 2). One year after the disposal, the Zn, Pb, Ni, Mn, Fe and Cu contents decreased to pre-disposal levels. This reduction could be due to the dispersal of disposed sediment across the sea floor. The diversity, evenness and richness of the benthic fauna decreased significantly at the disposal site after disposal of the dredged sediment (Annex 4). One year after disposal the benthic fauna recovered, but did not yet reach the recorded pre-disposal state. However there is no reason to suggest that they will not recover eventually (Galway Bay, 2003). From past experience, it appears that there is ample time between two disposal activities (disposal once a decade) for a ‘healthy’ bottom environment to be re-established. For the next disposal operation, the receiving area in the most recent operation should be left fallow, and another part of the designated disposal area be used instead.

Stronkhorst *et al.* (2003) studied the environmental impact and recovery at two disposal sites (site North and Northwest) in the North Sea (The Netherlands) after long and interrupted disposal of large volumes of moderately contaminated dredged sediments from the port of Rotterdam. During the period of sediment disposal very few benthic invertebrates were found at the North site (significant lower species richness and relative abundance). Concentrations of Cd, Hg, PCBs, PAHs and TBT in the fraction < 63 µm from this site were 2-3 times higher than at the reference site (Annex 3). However, levels of TBT were four times higher than the target value for imposex (Oehlmann *et al.*, 2000).In tissue (*pyloric caeca*) of resident starfish *Asterias rubens*, residual levels of Hg, Zn, PCBs and dioxin-like activity were never more than twice those at the reference site. In four different bioassays on starfish tissues (DNA integrity, cytochrome P450 content, benzo(*a*)pyrene hydroxylase activity and acetylcholinesterase inhibition), the sediments showed no acute toxic effects.

Minor pathological effects were observed in resident dab *Limanda limanda*. Pathological effects were either within the range normally found in coastal waters of the Netherlands or differences were not significant and could not be related to causal constituents that are associated with the disposed sediment. One year after sediment disposal had ceased at the North site, a significant increase in the species richness and abundance of benthic invertebrates and a concomitant decrease in the fine sediment fraction of the seabed was observed (Annex 4).

At site Northwest Stronkhorst *et al.* (2003) determined the impacts of sediment disposal from the disposal site till 8 km eastwards. During the time of disposal, the species richness and abundance of benthic invertebrates declined over an area extending about 1-2 km eastwards (Annex 4). This is correlated with a shift in sediment texture from sand to silt. The contamination (Cd, Hg, PCBs, PAHs and TBT) of the fine sediment fraction (<63 µm) at the Northwest location doubled (Annex 3). Stronkhorst *et al.* (2003) concluded that marine benthic resources at and around the disposal sites have been adversely effected by physical disturbance (burial, smothering). However, no causal link could be established with sediment-associated contaminants from the dredged sediment. The results of the study of Stronkhorst *et al.* (2003) are described in more detail in Annex 5.1 (extended summary).

Lauwaert *et al.* (2004) summarized the effects of the disposal of dredged sediments at four disposal sites (Annex 2) in the Belgium Continental Shelf (BCS) (permission period 2002-2003). At disposal site S1 the clay/silt fraction is most likely increased by the disposal of the dredged sediments. No conclusions could be drawn on the impact on sedimentology for the other three disposal sites.

The disposed sediments at LO and ZO (Annex 3) are rich in clay and silt. These sites show strongly alternating community structures (macrobenthos) due to the dominance of the fine sediments. The diversity and the number of species (macrobenthos) at these sites is also lower compared with the reference sites (and other disposal sites)(Annex 4). Disposal site S1 and S2 are characterized by a lower clay/silt content, which might explain the more stable community structure of the macrobenthos and the higher diversity and higher number of species.

Fish are, contrary to macrobenthos, highly mobile. During dredged sediment disposal activities, fish leave the disposal site. The return of fish depends on the presence of food resources. No significant difference is observed in the fish quantities at the disposal sites and the reference sites. This is an indication for the low impacts of disposal activities on the fish quantities.

The PCB and OCP content of the sediment at the disposal sites and reference sites was determined. No difference between the disposal and reference sites was observed (Annex 3). The Cr, Ni, Cu and Pb content at disposal site ZO is increasing significantly (fraction <2mm). No relation could be established between this increase and dredging activities at the BCS.

Lauwaert *et al.* (2004) also studied the effects of dredged sediment disposal on the health of fish. No relationship could be established between the occurrence and distribution of fish diseases (a.o. infections, tumours and malformations) and activities of dredged sediment disposal at the BCS.

Leuchs *et al.* (2004) summarized the effects caused by disposal of dredged sediments on the benthic macrofaunal communities in German estuaries and embayments. Their summary is based on data and statistical analysis obtained from research reports from a.o Bundesanstalt für Gewässerkunde (BfG), Wasser- und Schiffahrtsambt Emden, Wilhelmshaven, Cuxhaven, Brunsbüttel and Hamburg. One of these reports is described in more detail in this underlying document (BfG 2001). The following effects were included in their study: changes in turbidity, grain size composition, availability of nutrients, hazardous substances, and oxygen concentration. In general, dredged sediment-disposal below about 0.5×10^6 m^3 showed no, or in case of very small disposal areas (<0.4 km^2) no clear evidence for disposal induced effects on the macrozoobenthos communities, while higher amounts of dredged sediments resulted in clear effect on the macrozoobenthos communities. In case effects were observed, these effects were restricted to the disposal site and surroundings only. Leuchs *et al.* (2004) also conclude that natural conditions like weather and commercial fishing activities in most cases cause larger changes in abundance and composition of the benthic fauna communities than the disposal of dredged sediments.

Witt *et al.* (2004) studied the impact of harbour sludge disposal on benthic macrofaunal communities in the Weser estuary (Germany) during an open-water disposal of about 710,000 m^3 of harbour sludge. The macrofaunal communities of four sites within the disposal area and five sites in a reference area were compared after discharge. Sediments at the disposal area and the reference area differed at all times. Sediments in the disposal area had a higher percentage of silt and organic matter, and a lower median particle size than those of the reference area (Annex 2 and 3). Witt *et al.* (2004) measured a significant loss of diversity and a decline in the abundance of several species in the disposal area (Annex 4). The number of species was reduced up to 50% and important habitat structures were absent at the disposal area (Annex 4). Only the number of robust and opportunistic species increased (favouring fine grained and nutrient rich sediments).

Bolam and Whomersley (2005) studied the impact of the disposal of uncontaminated dredged sediment (used for mudflat enhancement) on the macrofauna at three different locations in the maritime area of south-east England. They concluded that parameters as biomass and diversity at the disposal sites attained reference levels within 1 year (Annex 4). The macrofaunal community structure of all disposal sites however, failed to converge to the reference sites within 1 year (Annex 4). These differences are, according to the researchers, unlikely to have been resulted from sediment differences (between natural situation and deposited dredged sediment), but may reflect some underlying differences in environmental conditions between disposal and reference sites. Natural spatial variability may have prevented total convergence between the disposal site and the reference site. In other words, based on the employed monitoring approach no hard conclusions could be drawn on the impacts of the disposal of dredged sediment on the macrofaunal communities.

The results of the study of Bolam and Whomersley (2005) are described in more detail in Annex 5.2 (extended summary).

CEFAS (2005) determined the environmental impacts resulting from disposal of dredged material at the Rame head disposal site in S.W. England. Between 1976 and 2005 over 5 million tonnes of dredged sediments have been deposited at Rame Head.

Evidence from monitoring surveys indicates that physical, chemical and biological impacts are localised within, and close to the boundaries of the disposal site. Concentrations of a number of chemical contaminants (As, Zn, Cu, Cd, TBT, PCBs, PAHs) in sediments sampled from the disposal site are higher than in the surrounding area but are consistent with the quality of sediments suitable for disposal to sea. The exception to this is concentrations of PAHs, which are elevated at the disposal site and in the surrounding area. The source of the PAH contamination is unknown. The biological communities are typical of those inhabiting comparable sediments elsewhere in the vicinity, both within and outside the disposal site. There is evidence of a stable community structure away from disposal operations. A more variable faunal community exists within and close to the disposal site, which is expected from the highly variable nature of the sea floor, but could also be due in part to the disposal regime. The benthic community structure shows no evidence of impacts of contaminants. Along the likely fine-sediment transport path, where any impacts might be expected, the communities show high species diversity. Furthermore, ecotoxicological testing of sediments, performed using bioassay techniques, provides strong supporting evidence for the absence of any contaminant induced impacts on marine animals.

CEFAS (2006) conclude that enhanced levels of certain contaminants (As, Zn, Cu, Cd, TBT, PCBs, PAHs) in the sediments and the variability in the benthic community at and near the disposal site may be attributable, at least in part, to the disposal operation. Such impacts are however within the range of the expected physical, chemical and biological disturbances from the disposal of dredged sediments and are regarded as acceptable in the light of the requirement for such disposal.

Lauwaert *et al.* (2006) summarized the effects of the disposal of dredged sediments at five disposal sites (Annex 2) in the Belgium Continental Shelf (BCS) (permission period 2004-2006). The silt/clay fraction at disposal site LS1 is highly variable in space and time. This variation is most likely caused by the local impact of the disposal of dredged sediment. The sediment composition of the other disposal sites, with the exception of LS2, is also highly variable. This is probably caused by the variable composition of the disposed dredged sediments in combination with the random distribution of the disposed dredged sediment.

Based on the results of an intensified sampling campaign Lauwaert *et al.* (2006) conclude that the disposal of dredged sediment at disposal site LS1 has a significant impact on the macrobenthos. Density, species richness and diversity of the macrobenthos at this site is significantly lower compared to the reference sites (Annex 4). The impact of dredged sediment disposal on the macrobenthos at the other for disposal sites is less clear, most likely due to the limited sampling campaign at these sites (Annex 4). The PCB, OCP. PAH, heavy metal and organotin content of the sediment at the disposal sites and reference sites was determined. No difference between the disposal and reference sites was observed (Annex 3).Two biochemical indicators (S.A. EROD and S.A. GSH-t) for pollution were determined. No relationship between the biochemical indicators and disposal activities was observed.

Lauwaert *et al.* (2006) also studied the effects of dredged sediment disposal on the health of fish. No relationship could be established between the occurrence and distribution of fish diseases (a.o. infections, tumours and malformations) and activities of dredged sediment disposal at the BCS.

Powilleit *et al.* (2006) studied the impacts of an incidental disposal of uncontaminated dredged sediment on the macrofaunal community in Mecklenburg Bay (Germany). They observed a clear deterioration in the local benthic fauna two weeks after the disposal. However, shortly after the disposal event survival and/or re-colonization was on a remarkable high level. Within two years after disposal, additional new recruitment processes caused an almost complete recovery of the benthic vertebrates in terms of total densities, species richness and total biomass. The macrofaumal community structure however, was still different after 2 years (Annex 4). Powilleit *et al.* (2006) postulate that these differences are due to inter-annual variability in regional benthic recruitment processes in the study area rather than to the disposal of the dredged sediment.

Somerfield *et al*. (2006) used a phylum-level meta-analysis approach to determine the relative impacts of dredged sediment relocation in the coastal environment. According to the researchers, the results of individual studies (as described above) are limited in terms of predicting the likely impacts of new disposal operations. By combining data from a variety of studies, a simultaneous comparison can be made, which can result in the detection of general trends. Based on a phylum-level meta-analysis applied to 192 samples from a variety of dredging disposals and relocation sites around the coast of England and Wales, they conclude that dredgings disposal has two contrasting impacts on benthic communities. One, associated with organic enrichment, leads to communities dominated by annelids and nematodes. The other, associated with intense physical disturbance, favours large motile or armoured forms, such as bivalve molluscs and crustaceans. Based on their method Somerfield *et al.* (2006) conclude that most samples were only moderately disturbed to undisturbed.

In recent years, dredged sediment has become regarded as a potential resource and used to create and/or improve intertidal habitats (i.e. beneficial use) (Bolam *et al*., 2006; Schratzberger *et al.*, 2006; Widdows *et al.*, 2006). Bolam *et al.* (2006) investigated the long-term (42 months post-recharge) macro- and meiofaunal recolonization processes of a beneficial use-scheme in south-east England Titchmarsh Marina). Although the material recharged (disposed) at Titchmarsh Marina was very fluid-like (92% water), the sediment rapidly dewatered and rich and diverse macro- and meiofaunal assemblages were present within 3 months. Significant differences in sediment properties and community structures between the recharge site and the reference sites were observed in time (Annex 4). The researchers ascribe the differences not to adverse effects by the recharge but by the inherent spatial differences in one or more influential physical properties (a.o. sediment texture). Therefore no attempt has been made to assess the extent to which the recolonising communities can be described as recovered . Bolam *et al.* (2006) propose that assessing recovery of a beneficial use scheme should be undertaken using pre-defined criteria in addition to comparisons with a reference site. The results of the study of Bolam *et al.* (2006) are described in more detail in Annex 5.3 (extended summary).

Van Dalfsen and Lewis (2006) studied the recovery of the macrofaunal community after disposal of dredged sediment at disposal site ‘Verdiepte Loswal’ in the Netherlands. The ‘Verdiepte Loswal’ is a 10 m deep excavation pit that is filled with dredged sediment. Disposal of dredged sediment in this pit ended in May 2004. In August 2006 Van Dalfsen and Lewis (2006) determined the number of species, density, biomass, evenness, diversity and community structure of the macrofaunal community at the former disposal site. They compared this with a reference site nearby. Significant differences in sediment properties were observed. The reference site mainly consist of fine to coarse sand with a low silt/clay content (0,02%), whereas the disposal site consists of finer sediments with a silt/clay content varying from 2,9 to 36,6%.

Two years and 3 months after ending the disposal activities at ‘Verdiepte Loswal’ the macrofaunal community of the former disposal site still differs from the reference site. The reference location is characterized by lower number of species and different species compared to the species at the former disposal site. In addition, the species are also more evenly distributed at the reference location. Van Dalfsen and Lewis (2006) conclude that the differences in the macrofaunal community are related to the differences in the sediment composition (a.o grain size). According to the researchers, complete recovery will only take place if the sand content of the sediment at the former disposal site increases.

Alzieu (2005), published a multidisciplinary manual designed for scientists, engineers and managers working in marine port environments. Large amounts of sediments are dredged to maintain ships' access to ports and harbours. Dumping dredged waste at sea is only authorized if they have been shown to be harmless for the marine environment. Following a brief review of the geochemistry and toxicity of the various contaminants adsorbed on sediments, this manual proposes a holistic and integrated approach to assessing the environmental hazards related to dredged materials disposal at sea. The document also gives a detailed description of risk characterization methods based on chemical analysis (Géodrisk), along with selected toxicity assays used for sediment quality bioassessments as defined by the Index for evaluation of the coastal endofauna (I2EC) and active biomonitoring of dumping sites. This iterative and bottom-up approach, as applied in the French regulatory context, fulfils international recommendation requirements, and especially OSPAR Convention guidelines.

Alzieu, (2003) proposes a number of monitoring methods designed to verify the impact hypothesis and readjust it with any required corrective measures. Thus, the I2EC index (Index for Evaluation of the Coastal Endofauna) assigns a bio-quality score to each type of sediment. Monitoring procedures in the open sea, based on the bio-accumulation properties of bivalves, now make it possible to check the impacts of contaminants inputs. One chapter deals with monitoring of inputs from confined disposal on land. The book will provide port managers, scientific researchers and environmental engineers with a bottom-up monitoring approach in compliance with OSPAR guideline recommendations.

CETMEF, (2002). Recueil de textes pour l'établissement d'un document d'incidence, CETMEF edition BP 60039, 60321 Compiègne cedex. This book proposes a collection of French regulations and impact studies

Annex 2. General characteristics (country, location name, coordinates, OSPAR region, environment, salinity and natural sediment composition) of the reviewed maritime locations in the OSPAR region where dredged sediment has been disposed and monitored.

Country	Location name	Coordinates		OSPAR region	Environment	Salinity (‰)	Natural sediment composition[1)]				Reference
		N	**E**				**Water (%)**	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	
Belgium	Loswal Br. & W. S1 (S1)			II	High-energetic hydrodynamic environment, wind en tides dominated				4	198	Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. S2 (S2)			II					4	198	Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. Zeebrugge Oost (ZO)			II					4	198	Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. Oostende (LO)			II					4	198	Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. S1 (LS1)			II	High-energetic hydrodynamic environment, wind en tides dominated	30-34				170-200	Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. S2 (LS2)			II	High-energetic hydrodynamic environment, wind en tides dominated	30-34				170-200	Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. Zeebrugge Oost (LZO)			II		30-34				170-200	Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. Oostende (LO)			II		30-34				170-200	Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. Nieuwpoort (LN)			II		30-34				170-200	Lauwaert *et al*. 2006
Germany	Elbeastuar	53º53.23’	09º05.46’	II							Leuchs and Nehring (1996)

Country	**Location name**	**Coordinates**		**OSPAR region**	**Environment**	**Salinity (‰)**	**Natural sediment composition**[1)]				**Reference**
		N	**E**				**Water (%)**	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	
Germany	Ems-Ästuar			II	High-energetic hydrodynamic environment, wind en tides dominated High-energetic hydrodynamic environment, wind en tides dominated						BfG (2001)
Germany	Wurster Arm (Weser estuary)			II	Polyhaline brackish-water zone, strong tidal currents up to 1.5m/s	17-30 psu		2.6 OM	8.9 silt	100	Witt *et al.* (2004)
Germany	Mecklenburg Bay - Station 1	54º12.027’	11º54.160’	Baltic sea (near region II)	Non-tidal conditions, weather dependant variable water	13.5-18.8 psu	22		5		Powilleit *et al.* (2006)
Germany	Mecklenburg Bay - Station 2	54º12.118’	11º54.048’	Baltic sea (near region II)	Non-tidal conditions, weather dependant variable water	13.5-18.8 psu	22		5		Powilleit *et al.* (2006)
Netherlands	Oude Westereems			II				1.4 (’89) 1.7 (’90) OM	12.5 (’89) 15.0 (’90)		Kleef *et al.* (1992)
Netherlands	Dumping site North (N4)			II	High-energetic hydrodynamic environment, wind en tides dominated				2-10	± 300	Stronkhorst *et al.* 2003
Netherlands	Dumping site North (N53)			II					2-10	± 300	Stronkhorst *et al.* 2003
Netherlands	Dumping site Northwest			II		30			2-10	± 300	Stronkhorst *et al.* 2003
Netherlands	Verdiepte Loswal	569500 UTM X	5767900 UTM Y	II					0,02	268	Van Dalfsen and Lewis (2006)
United Kingdom	Westwick Marina	51º38.692’	00 º39.61’	II	Muddy channels in saltmarch system		62.2	2	91.2		Bolam and Whomersley (2005)

Country	**Location name**	**Coordinates**		**OSPAR region**	**Environment**	**Salinity (‰)**	**Natural sediment composition[1)]**				**Reference**
		N	**E**				**Water (%)**	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	
United Kingdom	Titchmarch Marina	51º51.763'	001º15.133'	II	Muddy channels in saltmarch system		73.6	2.78	98.7		Bolam and Whomersley (2005)
United Kingdom	North Shotley	51º57.973'	001º16.469'	II	Mudflat		50.8	1.27	91.7		Bolam and Whomersley (2005)
United Kingdom	Rame Head	50 18.900 N	04 16.490 W	II	Moderate energy regime						CEFAS (2005)
United Kingdom	Titchmarch Marina	51º51.763'	001º15.133'	II	Intertidal flats and saltmarches		49.6 - 75.2	1.8 - 3.0	58.2 - 90.8		Bolam *et al.* (2006)
Ireland	Galway Bay spoilgrounds	53º15.48'	09º00.37'	III							Galway Bay (2003)
United Kingdom	Liverpool Bay			III	Tidal currents 0.8m/s						Rees et al. (1992)

1) Often based on measurements on reference locations

Annex 3 General information (amount of sediment disposed, frequency of disposal, surface area of disposal site, period of disposal, thickness of disposed sediment layer and depth of disposal site) regarding disposal activities and composition (contaminants, water content, carbon content, silt/clay content and medium grain size) of the disposed dredged sediments in the OSPAR maritime area.

Country	**Location**	**Disposed dredged sediment**											**Reference**
		Amount (m^3)	**Frequency of disposal**	**Surface area (m^2)**	**Period of disposal**	**Thickness (cm)**	**Depth after disposal (m)**	**Contaminated[1)]**	**Water (%)**	**Carbon (%)**	**Silt/clay (%)**	**Medium grain size (μm)**	
Belgium	Loswal Br. & W. Nieuwpoort (LN)	>500000 tonnes	Yearly from 2003		'03 till at least '05			ND from reference site (OCPs / PCBs/PAHs/ organotin)					Lauwaert *et al.* 2006
Belgium	Loswal Br. & W. S1 (S1)	>120000000 tonnes	Yearly from 1991		'91 till at least '03			ND from reference site (OCPs / PCBs)			6	±192	Lauwaert *et al.* 2004
Belgium	Loswal Br. & W. S2 (S2)	>35000000 tonnes	Yearly from 1991		'91 till at least '03			ND from reference site (OCPs / PCBs)			3	207	Lauwaert *et al.* 2004
Belgium	Loswal Br.& W.Zeebrugge Oost (ZO)	>75000000 tonnes	Yearly from 1991		'91 till at least '03			OCPs and PCBs: ND from reference site. Cr, Cu, Ni and Pb at ZO increasing.			84	42	Lauwaert *et al.* 2004
Belgium	Loswal Br. & W. Oostende (LO)	>25000000 tonnes	Yearly from 1991		'91 till at least '03			ND from reference site (OCPs / PCBs)			24	142	Lauwaert *et al.* 2004

Belgium	Loswal Br. & W. S1 (LS1)	>130000000 tonnes	Yearly from 1991		‘91 till at least ‘05			ND from reference site (OCPs / PCBs/PAHs/ zware metalen)					Lauwaert *et al.* 2006
Belgium	Loswal Br. & W. S2 (LS2)	>37000000 tonnes	Yearly from 1991		‘91 till at least ‘05			ND from reference site (OCPs / PCBs/PAHs/ zware metalen/organotin)					Lauwaert *et al.* 2006
Belgium	Loswal Br. & W. Zeebrugge Oost (LZO)	>80000000 tonnes	Yearly from 1991		‘91 till at least ‘05			ND from reference site (OCPs / PCBs/PAHs/ zware metalen/organotin)					Lauwaert *et al.* 2006
Germany	Elbeastuar	3100000-6200000 per year			from 1895 till at least 1995								Leuchs and Nehring (1996)
Germany	Ems- Ästuar	±6000000 per year			1999-2000			No (As, Cd, Ni, Pb, Zn, Cr, Hg, PCBs, PAHs, TBT). Yes slightly (Cu, HCB).					BfG (2001)
Germany	Wurster Arm (Weser estuary)	550000 per year	Weakly to monthly	± 500000	Form 1960 till at least 1997	6.5							Witt *et al.* (2004)
Germany	Mecklenburg Bay - Station 1	2800	Once	250000	June 2001	150	19						Powilleit *et al.* (2006)
Germany	Mecklenburg Bay - Station 2	2400	Once	250000	June 2001	150	19						Powilleit *et al.* (2006)
Ireland	Galway Bay spoilgrounds	80000	Once per decade	405000	Aug-Sep 2001			Yes (Zn, Pb, Ni, Mn, Fe and Cu)					Galway Bay (2003)

Country	Location	Disposed dredged sediment											Reference
		Amount (m^3)	**Frequency of disposal**	**Surface area (m^2)**	**Period of disposal**	**Thickness (cm)**	**Depth after disposal (m)**	**Contaminated[1)]**	**Water (%)**	**Carbon (%)**	**Silt/clay (%)**	**Medium grain size (μm)**	
Netherlands	Dumping site North (N4)		Yearly (till 1996)		Till 1996			Yes (TBT, PCBs, PAHs, Cd and Hg in <63mm) factor 2-3			40	100	Stronkhorst *et al.* 2003
Netherlands	Dumping site North (N53)	8100000	Yearly (till 1996)		Till 1996		10-12	Yes (TBT, PCBs, PAHs, Cd and Hg in <63mm) factor 2-3					Stronkhorst *et al.* 2003
Netherlands	Dumping site Northwest	8200000	Yearly (from 1996)	250000	From 1996	300	18						Stronkhorst *et al.* 2003
Netherlands	Oude Westereems	544000 (1989), 850000 (1990)	Yearly	157000	In 1989 and 1990	90	10-15						Kleef *et al.* (1992)
Netherlands	Verdiepte Loswal. Put 1. Location 23				Till May 2004	1000 (in a pit with a depth of 1000 cm)							Van Dalfsen and Lewis (2006)
Netherlands	Verdiepte Loswal. Put 1. Location 24				Till May 2004	1000 (in a pit with a depth of 1000 cm)							Van Dalfsen and Lewis (2006)
United Kingdom	Westwick Marina		Once			60-80	1.2-1.3	Not (metals and TBT)	91.2	1.3	94.5		Bolam and Whomersley (2005)
United Kingdom	Titchmarch Marina		Once			60-80	1.5-1.9	Not (metals and TBT)	91.7	1.2	95		Bolam and Whomersley (2005)

Country	**Location**	**Disposed dredged sediment**											**Reference**
		Amount (m^3)	**Frequency of disposal**	**Surface area (m^2)**	**Period of disposal**	**Thickness (cm)**	**Depth after disposal (m)**	**Contaminated[1)]**	**Water (%)**	**Carbon (%)**	**Silt/clay (%)**	**Medium grain size (μm)**	
United Kingdom	North Shotley		Once			60-80	1.4-1.5	Not (metals and TBT)	60.3	1.5	93.7		Bolam and Whomersley (2005)
United Kingdom	Rame Head	>5000000 tonnes	Yearly from 1976		1976-2005		18-38 m below CD	D from reference (As, Zn, Cu, Cd, TBT, PCBs, PAHS). ND from reference (Ni, Pb, Hg, Cr, BFR).			>70		CEFAS (2005)
United Kingdom	Liverpool Bay	3000000 tonnes	Continuously from 1955 to at least 1988				10				70% sand 30% mud		Rees *et al.* (1992)
United Kingdom	Titchmarch Marina		Once		May, june 2001	65-80		Not (trace metals, TBT)	92	1.2	95		Bolam *et al.* (2006)

1)ND=Not different

Annex 4. Measured and/or observed effects on sediment composition (medium grain size and content of water, carbon and silt/clay) and fauna (diversity, evenness, biomass, total density, total individuals, number of species, community structure and abundance) - in time and space – at disposal sites in the OSPAR regions, as found in the literature.

Country	**Location**	**Time after disposal (weeks)**	**Distance from disposal site (km)**	**Effects-Sediment composition**				**Effects-Fauna**									**Reference**
				Water (%)	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	**Type[2)]**	**Diver-sity[3)]**	**Even-ness**	**Bio-mass**	**total density**	**Total indivi-duals**	**Number of species / species richness**	**Com-munity structure**	**(Relative) abundan-ce**	
Belgium	Loswal Br. & W. S1 (S1)	Continuing disposal	0					MA	ND					ND	ND		Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. S2 (S2)	Continuing disposal	0					MA	ND					ND			Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. Zeebrugge Oost (ZO)	Continuing disposal	0					MA	D (lower)					D (lower)	D (more variable)		Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. Oostende (LO)	Continuing disposal	0					MA	D (lower)					D (lower)	D (more variable)		Lauwaert *et al*. 2004
Belgium	Loswal Br. & W. S1 (LS1)	Continuing disposal	0					MA	D (lower)			D (lower)		D (lower)			Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. S2 (LS2)	Continuing disposal	0					MA	ND			ND		ND			Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. Zeebrugge Oost (LZO)	Continuing disposal	0					MA	ND			ND		ND			Lauwaert *et al*. 2006
Belgium	Loswal Br. & W. Oostende (LO)	Continuing disposal	0					MA	ND			ND		ND			Lauwaert *et al*. 2006

Country	**Location**	**Time after disposal (weeks)**	**Distance from disposal site (km)**	**Effects-Sediment composition**				**Effects-Fauna**									**Reference**
				Water (%)	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	**Type**[2)]	**Diver-sity**[3)]	**Even-ness**	**Bio-mass**	**total density**	**Total indivi-duals**	**Number of species / species richness**	**Com-munity structure**	**(Relative) abundan-ce**	
Belgium	Loswal Br. & W. Nieuwpoort (LN)	Continuing disposal	0					MA	ND			ND		ND			Lauwaert *et al.* 2006
Germany	Elbeastuar	continuous disposal	0-1					MA				↓	↓	↓			Leuchs and Nehring (1996)
Germany	Ems- Ästuar	continuous disposal						MA							D	ND	BfG (2001)
Germany	Wurster Arm (Weser estuary)	Several weeks	0		8.1 OM	41.1 silt	80	MA					↓	↓	D (more variable)		Witt *et al.* (2004)
Germany	Mecklenburg Bay - Station 1	2	0					MA	↓		↓	↓		↓	NE		Powilleit *et al.* (2006)
Germany	Mecklenburg Bay - Station 2	2	0					MA	↓		↓	↓		↓	NE		Powilleit *et al.* (2006)
Germany	Mecklenburg Bay - Station 1	54	0					MA			ND	ND		NC	NE		Powilleit *et al.* (2006)
Germany	Mecklenburg Bay - Station 2	54	0					MA			ND	ND		ND	NE		Powilleit *et al.* (2006)
Ireland	Galway Bay spoilgrounds	8			↑				↓	↓				↓			Galway Bay (2003)
Ireland	Galway Bay spoilgrounds	52			ND				↓ BR	↓ BR				↓ BR			Galway Bay (2003)

Country	**Location**	**Time after disposal (weeks)**	**Distance from disposal site (km)**	**Effects-Sediment composition**				**Effects-Fauna**									**Reference**
				Water (%)	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	**Type[2)]**	**Diver-sity[3)]**	**Even-ness**	**Bio-mass**	**total density**	**Total indivi-duals**	**Number of species / species richness**	**Com-munity structure**	**(Relative) abundan-ce**	
Netherlands	Oude Westereems	0-52	0		1.5 ('89) 3.8 ('90) OM	13.9('89) 32.2('90)		MA				↓		↓ (50%)			Kleef *et al.* (1992)
Netherlands	Dumping site North (N4)	0	0					MA + MEG						↓		↓	Stronkhorst *et al.* 2003
Netherlands	Dumping site North (N53)	0	0					MA + MEG						↓		↓	Stronkhorst *et al.* 2003
Netherlands	Dumping site North (N4)	52	0			5	200	MA + MEG					ND		ND		Stronkhorst *et al.* 2003
Netherlands	Dumping site North (N53)	52	0					MA + MEG						ND		ND	Stronkhorst *et al.* 2003
Netherlands	Dumping site Northwest	52	0					MA + MEG			↓ MA			↓		↓	Stronkhorst *et al.* 2003
Netherlands	Dumping site Northwest	52	1			30	<200	MA						↓ MA ND MEG		↓ MA ND MEG	Stronkhorst *et al.* 2003
Netherlands	Dumping site Northwest	52	2			16		MA			↓ MA			ND		ND	Stronkhorst *et al.* 2003
Netherlands	Dumping site Northwest	52	3			<5		MA						ND		ND	Stronkhorst *et al.* 2003

Country	**Location**	**Time after disposal (weeks)**	**Distance from disposal site (km)**	**Effects-Sediment composition**				**Effects-Fauna**									**Reference**
				Water (%)	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	**Type[2)]**	**Diver-sity[3)]**	**Even-ness**	**Bio-mass**	**total density**	**Total indivi-duals**	**Number of species / species richness**	**Com-munity structure**	**(Relative) abundan-ce**	
Netherlands	Dumping site Northwest	52	5			<5		MA						ND		ND	Stronkhorst *et al.* 2003
United Kingdom	Liverpool Bay		0		↑			MA									Rees *et al.* (1992)
United Kingdom	Westwick Marina	52	0	66.5	2.7	89.5		MA	ND	ND	ND		ND	↓	NE		Bolam and Whomersley (2005)
United Kingdom	Titchmarch Marina	52	0	60.6	1.58	88.2		MA	ND	ND	ND		↓	↓	NE		Bolam and Whomersley (2005)
United Kingdom	North Shotley	52	0	55.2	1.37	93.8		MA	ND	ND	ND		ND	ND	NE		Bolam and Whomersley (2005)
United Kingdom	Rame Head	Continuing disposal	0					MA						↓			CEFAS (2005)
United Kingdom	Titchmarch Marina	13	0	62.3	1.6	97.4		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	26	0	58.8	1.6	95.9		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	52	0	60.6	1.6	88.2		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	78	0	62.1	1.3	95.3		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	104	0	57.8	1.3	89.8		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	130	0	59.5	1.3	92.5		MA	↓					↓	D	↓	Bolam *et al.* (2006)

Country	**Location**	**Time after disposal (weeks)**	**Distance from disposal site (km)**	**Effects-Sediment composition**				**Effects-Fauna**									**Reference**
				Water (%)	**Carbon (%)**	**Silt/clay (%)**	**medium grain size (μm)**	**Type[2)]**	**Diver-sity[3)]**	**Even-ness**	**Bio-mass**	**total density**	**Total indivi-duals**	**Number of species / species richness**	**Com-munity structure**	**(Relative) abundan-ce**	
United Kingdom	Titchmarch Marina	156	0	63.2	1.6	82.9		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	208	0	62.1	1.3	82.1		MA	↓					↓	D	↓	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	13	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	26	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	52	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	78	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	104	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	130	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	156	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
United Kingdom	Titchmarch Marina	208	0					ME	ND					ND	D	ND	Bolam *et al.* (2006)
Netherlands	Verdiepte Loswal. Put 1. Location 23	117	0			2.9	208	MA	D	D	D			D	D		Van Dalfsen and Lewis (2006)
Netherlands	Verdiepte Loswal. Put 1. Location 24	117	0			36.6	98.6	MA	D	D	D			D	D		Van Dalfsen and Lewis (2006)

2) MA= Macrofauna; MEG=Megafauna; ME=Meiofauna

3) ND= Not different (compared to reference sites or compared to situation before disposal); D=Different (compared to reference sites or compared to situation before disposal); NE= No equilibrium; ↓ decrease (significant); ↓ BR decrease (significant) but recovering compared to previous monitoring period; ↑ increase (significant).

Annex 5.1 Disposal sites Loswal North and Loswal Northwest at the Netherlands Continental Shelf[9]

Introduction

Rotterdam, the foremost port in the Netherlands, requires intensive maintenance dredging that yields 15-18 million m^3 of sludge annually. The material to be dredged is sand and silts originating from the southern part of the North Sea and alluvial sediments from Rhine and Meuse. The dredged spoils consist predominantly of slightly contaminated marine silt from the mouth of the port plus smaller volumes of moderately contaminated sediments from the inner harbour. The dredged sediments that meet the threshold values for open water disposal have mainly been disposed at two locations in the North sea. These locations are called the North site and the Northwest site. Both sites are located in shallow coastal waters and near where the Rhine and Meuse discharge into the North Sea through the Nieuwe Waterweg (figure 2). This is a high-energetic hydrodynamic environment in which wind and tides transport most of the disposed dredged silty sediment northwards.

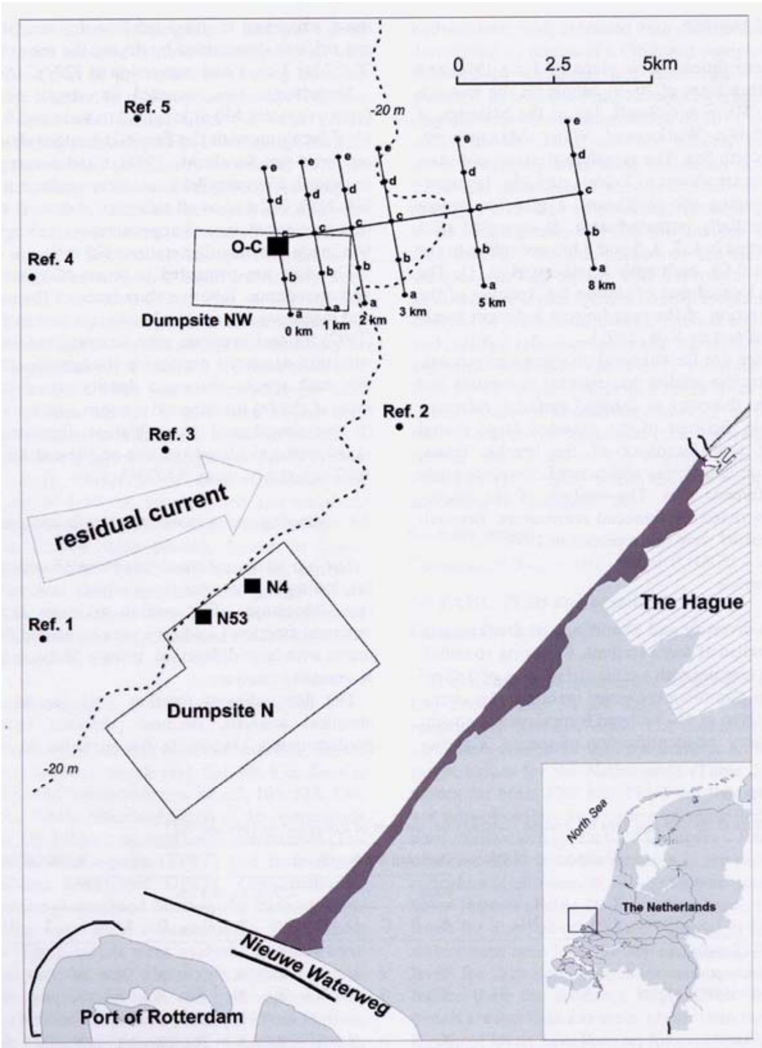

Figure 2. Sampling locations for dredged sediments from the port of Rotterdam at the former dumping site North and the new dumping site Nortwest. Also indicated are the sampling grid east of dumping site Northwest and reference sites 1-5 (Source: Stronkhorst *et al.*, 2003).

9 Sources: RIKZ (2002); Stronkhorst *et al*. (2003)

Between the early 1960s and June 1996 the harbour sediments were disposed at the North site. However it appeared that a substantial part of the disposed sediments re-entered the port via density currents through the salt wedge in the Nieuwe Waterweg. A new disposal site (Northwest) was therefore selected away from the current approximately 10 km from the North site (figure 2). The Northwest site came into use in July 1996 at the time that the disposal activities at site North were terminated.

RIKZ (2002) and Stronkhorst et al. (2003) reported the results of a study on the environmental impacts of the disposal of dredged sediment at the North and Northwest site in the North Sea. This study mainly focussed on the impacts on sediment chemistry, ecotoxicity, biomarker response, local benthic community structure, the recovery after disposal has ceased and re-entering rate of the disposed dredged sediments. Only near-field (several kilometres around the disposal locations) and short-term effects (a 1-year period) were determined. Long-range and multi-annual changes were not directly addressed. The study was a compliance monitoring exercise to provide feedback to be used when granting licences for the disposal of contaminated dredged sediments.

Materials and methods

General physical conditions at dumping sites

During the 12 months prior to the sampling at North in June 1996, approximately 8.1 million m^3 dredged material was disposed at location N53 (figure 2), where the water is 10-12 meter deep. The disposal of dredged sediment at North ceased in June 1996.

The disposal of dredged sediment at the new site, Northwest, took place between July 1996 and July 1997. In total 8.2 million m^3 was disposed in an area of 500 × 500 m around station 0-C (figure 2). The disposal reduced the water depth from 21 to 18 m at the actual disposal location 0-C and by approximately 0.5 m in an area 500-1000 from the disposal location.

During the surveys, the water temperature, salinity and dissolved oxygen concentration were approximately 16ºC, 30‰ and 100% respectively.

Sampling

The fieldwork was mainly conducted between 1996 and 1999. The sample locations are shown in figure 2. At disposal site North two locations were sampled, N53 and N4. At disposal site Northwest, a grid was sampled over the potentially impacted area. It consisted of 6 tracks (numbered 0, 1,2,3,5 and 8 km according to the distance from 0-C), each with 5 stations (figure 2). In addition five references sites were selected for sampling and analysis (figure 2).

Measurements

The field and laboratory work included a suite of measurements. General characteristics including sediment grain size distribution and biomass were determined. The abundance and species richness were quantified for two size classes of benthic invertebrates, namely smaller (that stays on a 1 mm sieve) and juvenile fauna (henceforth referred to as macrofauna) and the larger (that stay on a 1 cm sieve) infaunal and epifaunal species (referred to as megafauna). The degree of contamination at the disposal locations was analysed both in the fine sediment fraction (<63 μm) and in tissue of local starfish. Acute sediment ecotoxicity was measured with four in vivo bioassays. Sub-cellular toxicological effects were determined using four biomarkers in starfish tissue (*pyloric caeca*): 1) DNA integrity to assess the overall exposure to mutagenetic compounds, 2) the content of cytochrome P450 isozymes, a part of the monooxygenase enzyme system that can be affected by xenobiotics, 3) the activity of benzo(*a*)pyrene hydroxylase (BPH) as a measure of monooxygenese activity and 4) the inhibition of the enzyme acetylcholinesterase (AChE) by possible exposure to organophosphorus insecticides. The exposure to compounds with a dioxin-like mode of action was measured with the DC-CALUX in vitro bioassay. The levels of metals present in the dredged sediments were too low to justify applying a specific biomarker to examine the effects caused by trace metals (e.g. metallothioneins). Fish pathology was studied in the flat fish 'dab'. Finally, the percentage of disposed dredged sediments that re-entered the North Sea (left the disposal areas) was determined based on computer simulations.

Results and discussion

North site

During the period of disposal, very few benthic invertebrates were found at the North site. Concentrations of cadmium, mercury PCBs, PAHs and TBT in the fine sediment from this site were 2-3 times higher than at the reference site(s). In four different bioassays with marine invertebrates the disposed sediments did not show any acute toxic effects. In tissue (*pyloric caeca*) of resident starfish, residual levels of mercury, zinc, PCBs and dioxin-like activity were never more than twice those at the reference site. Four different biomarkers (DNA integrity, cytochrome P450 content, benzo(α)pyrene hydroxylase activity and acetylcholinesterase inhibition) were used on starfish tissue, but no significant differences were found between North and the reference site(s). Minor pathological effects were observed in resident dab.

One year after disposal with dredged silty harbour sediments has ceased, the sediment texture had recovered rapidly, almost attaining the reference conditions. The rapid distribution of the fine sediment layers from the disposal site to other areas of the North Sea was achieved by dynamic transport along the seabed in this shallow and high-energetic environment. The percentage of sediments re-entering the North Sea (leaving the disposal site) in 1999 was calculated to be 44 ± 22%. Within a short period, the benthic fauna – particularly the crustaceans and polychaetes – had substantially recolonized disposal site North. This recolonization of the benthic invertebrates is the result of the ending of physical disturbances (burial, smothering) and a decrease of silt content of the sediments. The observed rapid recolonization is typical of dynamic coastal areas and has been reported in a number of cases where dumping sites have been abandoned.

Approximately two years after disposal (in 1998) the abundance, species richness and biomass were even higher than at the reference locations. This indicates that a stable end-situation was not reached yet. This development was most likely related to the presence of both coarse and fine sediment, which can lead to a great(er) number of species.

In 1999, the difference in abundance, species richness and biomass decreases a little, approaching the conditions at the reference locations.

Although the silt content and median grain size of the bed floor sediment recovered, the vertical sedimentological structure did not and it is suspected that this will not recover in a short time span.

Northwest site

After 8.2 million m^3 of moderately contaminated dredged material had been dumped at the 'new' disposal site Northwest, the benthic fauna at the actual disposal location (0-C) had vanished. And in the same 2 km zone it was suppressed (decrease in species richness and abundance). The species of megafauna sampled did not show a significant decrease outside the disposal location 0-C. The smaller macrofauna sampled revealed a significant decrease as far as 1-2 km east of the actual disposal location 0-C to a level of species richness and bioassays of the macrofauna at site North in the previous year.

After disposal of dredged sediments at the Northwest site (in 1997), the concentrations of PCBs, PAHs. TBT, cadmium and mercury in the fine sediment fraction (<63 μm) had approximately doubled and were close to the concentrations measured in 1996 at the disposal site North.
Ecotoxicological test were not performed at the Northwest site, since no ecotoxicological effects were measured at the old disposal site, the North site.

In 1999, the species richness and abundance was still very low. There were no sings of recovery and no estimation of the recovery time could be made based o the limited number of data in time.

One of the reasons to close the disposal site North, was the high percentage of disposed dredged sediment that left the disposal site (re-entering the North Sea). This percentage is of great economical importance, since it determines the distance between dredging locations and disposal sites and thus shipping costs. A re-entering percentage of 17,4% in 1999 is calculated based on model calculations.

Conclusions

It is concluded that marine benthic resources at and around the disposal sites (North and Northwest) have been adversely affected by physical disturbance (burial, smothering). However, marine benthic resources were not affected (no toxicological effects) by the higher concentrations of cadmium, mercury PCBs, PAHs and TBT in the fine sediment of the disposed dredged sediments. The researchers advise to spread the dredged sediment in thin(ner) layers in the future. The thickness should be coupled to the maximum depth at which bottom dwellers can survive (escape by vertical migration).

Annex 5.2 Mudflat enhancement sites Westwick Marina, Titchmarsh Marina and North Shotley – United Kingdom Continental Shelf[10].

Introduction
The concern on the effects of the disposal of dredged sediments at sea has resulted in a greater emphasis on the relocation of dredged sediments (fine-grained) in such a way to derive environmental benefits. As a result a number of 'beneficial use' options have been developed whereby the dredged sediment is regarded as a potential resource and used to recharge or recreate intertidal habitats. In the US, dredged sediments have been shown to successfully create new mudflats and saltmarches, which ultimately function like natural systems. In the OSPAR maritime area (i.e Contracting Party UK), concerns over the eventual fate of the dredged sediments and the ecological consequences of placing fine-grained sediments onto intertidal habitats have limited this practise to small-scale field trials.

Bolam and Whomersley (2005) studied the invertebrate communities at a number of recharge and reference stations within three comparable beneficial use schemes in south-east England (OSPAR region II) one year after recharge. The main objectives of the Bolam and Whomersley (2005) study were to compare,

- (i) univariate community attributes (total individuals, total species, diversity, evenness and total biomass);
- (ii) species composition;
- (iii) trophic composition within and between schemes, and
- (iv) to propose factors responsible for any differences.

Materials and methods
Study sites
The three beneficial use schemes investigated during the present study were at Westwick Marina (WW), Titchmarsh Marina (TM) and North Shotley (NS) (see figure 3). Each scheme consisted of either a mudflat (NS) or muddy channels within a saltmarsh system (WW and TM) recharged with 60–80 cm (vertical overburden) of fine-grained, maintenance dredged sediments. The sediments recharged were considered uncontaminated in terms of metals and TBT. The resulting tidal height of each scheme was below the limits of saltmarsh plant colonisation (i.e., 2.1 mOD) and, consequently, high-level mudflats were the most appropriate references to assess faunal recovery. Reference sites were located as near as possible to the recharge area without being impacted by the recharge process itself.

Sampling and measurements
For each beneficial use scheme, three stations were positioned within the recharge (figure 3) and reference sites (figure 3). At each station, replicate macrofaunal samples (n = 3) were taken using a 0.01 m^2 perspex corer to a depth of 15 cm. On return to the laboratory the macrofaunal samples were processed. The top 3 cm of the sediment at each station was sampled for sediment analyses (water content, organic content and particle size distribution analyses. Replicate (n = 3) redox potential profiles (1, 2 and 4 cm) were measured at each station.

Data analysis
The sediment and macrofaunal data collected one year post-recharge were analysed using statistical techniques (both univariate and multivariate data analysis). Univariate techniques were used to test for differences in recharge and reference sites for each scheme. Relationships between total individuals and number of species with sediment variables of the recharge stations were investigated using univariate techniques. Multivariate analyses were carried out to assess (dis)similarities in community structure between recharge and reference stations and between schemes. Data analysis is described in more detail by Bolam and Whomersly (2005).

[10] Source: Bolam and Whomersley (2005)

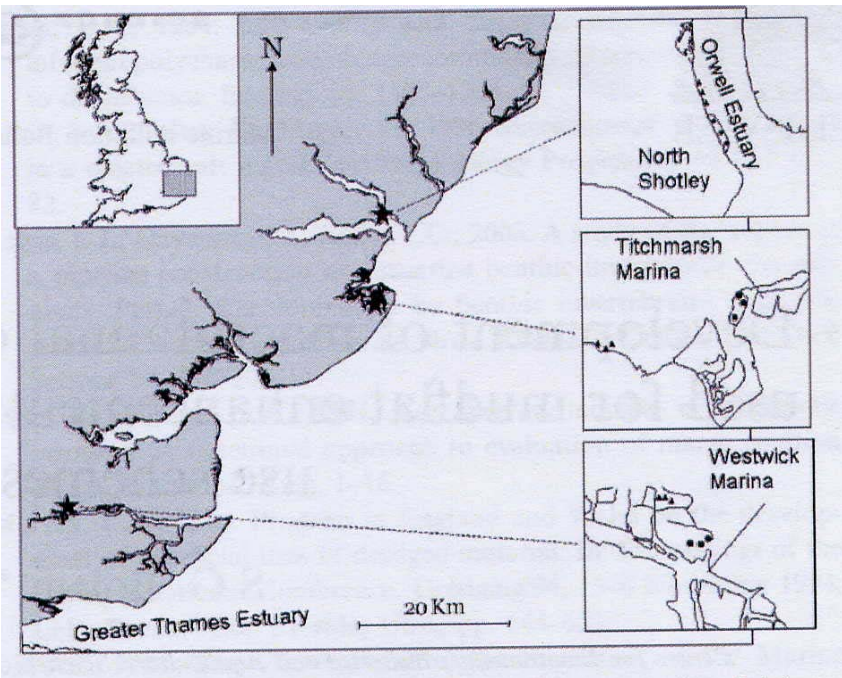

Figure 3. Map showing relative locations of the three beneficial use schemes and the positions of the recharge (circles) and reference (triangles) stations (Source: Bolam and Whomersley (2005)).

Results

Sediment data

The material recharged at WW and TM had markedly greater water contents relative to the sediments recharged at NS. However, these sediments greatly dewatered to give significantly lower water contents than reference sediments at TM after one year. The organic carbon and silt/clay contents were similar in all three dredged materials. The silt/clay contents were similar between the schemes, both in the recharge and reference sites, however, organic carbon content of the recharged sediments were significantly lower than reference sediments after one year. There were no significant differences in the redox potential profiles between recharge and reference sites except a significantly increased redox potential at 1 cm sediment depth at TM recharge site.

Invertebrate data

Total individuals, total species, diversity, evenness and total biomass (univariate indices of macrofauna) show various degrees of convergence to reference values (see also Annex 3); while there were no significant differences for diversity, evenness and biomass for any scheme, total individuals (TM) and number of species (WW and TM) were significantly lower in the recharge than reference sites.

Multivariate analyses reveal that the reference communities were very dissimilar between schemes, and secondly, that the reference communities exhibited large within-site variability, i.e., between stations. Furthermore, large differences between the recharge and reference communities for each scheme are observed. The invertebrate communities of recharge sites (after one year) are significantly different from reference sites for each scheme. The results also indicate that there are significant differences between the recharge communities of each scheme from each of the other schemes. The same holds true for the reference communities.

In total, six trophic groups were sampled throughout the study of Bolam And Whomersley (2005), most of these being present in all recharge and reference communities (except the reference site at NS which was composed of three trophic groups only). The recovery of trophic groups varied between schemes. For example, while the relative proportions of trophic groups were similar between recharge and reference communities at WW, the sub-surface deposit feeder numerical dominance at TM was replaced by a grazer dominance at the recharge site. Although the high proportion of surface deposit feeders at NS reference site had not established at the recharge site, the recharge site exhibited a larger number of trophic groups.

The relationships between univariate indices (total individuals and number of species) and sediment variables (water, carbon and silt/clay contents, 1, 2 and 4 cm redox potential) were investigated using a correlation approach. Total individuals were negatively correlated with 4 cm redox potential and number of species negatively correlated with % silt/clay.

Discussion

Significant differences in sediment properties (water, carbon and silt/clay contents, 1cm redox potential) between recharge and reference sites existed only at TM, the scheme which displayed the greatest deviation from the reference site for univariate indices (significant difference for total individuals and species) and multivariate community structure (84.5% dissimilarity from reference site community). This may indicate that sediment differences may have been responsible for the lack of convergence of recharge site communities with reference communities at TM. However, although 4cm redox potential and % silt/clay were significantly correlated with total abundance and number of species, respectively, sediment variables exhibited poor correlation with multivariate community structure of the recharge sites. According to Bolam and Whomersley (2005) further investigation is needed to determine how critical the degree of similarity of recharged sediments to those in reference areas is for invertebrate recolonization.

Although most univariate indices recovered within 12 months, macrofaunal community structure of the recharge sites failed to converge to those of reference sites for all three schemes. These differences are unlikely to have resulted from sediment differences but may reflect some underlying differences in environmental conditions between recharge and reference sites. The communities recolonising the dredged sediments developed (most likely) towards those collected at the same stations prior to dredged material disposal rather than those of the reference site. This suggests that natural spatial variability in response to underlying physical and/or biological processes may prevent total convergence between recharge and reference sites. Bolam and Whomersley (2005) postulate that a more functional-based approach rather than a community structure approach would possibly be more suitable for future studies on the impacts of dredged sediments used for mudflat enhancement.

Conclusions

Bolam and Whomersley (2005) conclude that although univariate community attributes of the macrofaunal community (total individuals, total species, diversity, evenness and total biomass) mostly recovered within one year, differences in community structure still existed one year after disposal.

The differences in community structure of the macrofauna are, according to the researchers, unlikely to have been resulted from sediment differences (between natural situation and deposited dredged sediment), but may reflect some underlying differences in environmental conditions between disposal and reference sites. Natural spatial variability may have prevented total convergence between the disposal site and the reference site.

Nevertheless rich and diverse infaunal communities (although different from reference locations) have established on mudflats after recharge with dredged sediments. Bolam and Whomersley (2005) attribute the relative rapid recolonization partly to the high resilience of mudflat communities and to the similarity of the dredged sediments to reference sediments in terms of organic carbon and silt/clay content.

Annex 5.3 Mudflat enhancement site Titchmarsh Marina – United Kingdom Continental Shelf[11].

Introduction

Following concerns regarding dredged material disposal in the marine environment, there is now a greater emphasis on the relocation of such material to derive environmental or other benefits. One current practice is the use of dredged material for habitat creation and/or restoration (i.e., beneficial use): such sediments have been shown to successfully enhance or create new mudflats and saltmarshes. In the UK, however, concerns over the eventual fate of the material and the ecological consequences of placing fine-grained material onto intertidal habitats have limited this practice to small-scale field trials.

Bolam *et al.* (2006) have therefore examined, over a 42-month period, the development of macrofaunal and meiofaunal assemblages of a fine-grained beneficial use scheme (in the UK) in an attempt to identify the spatial and temporal patterns of community development and the factors affecting them. The temporal duration of this study allowed the assessment of both seasonal and inter-annual variation in recolonisation processes.

In their study Bolam et al . (2006) mainly focussed on the following questions:

1. How do attributes of meio- and macrofaunal assemblages develop in time and space on newly recharged sediments (in the Titchmarsh Marina)?
2. Are the temporal patterns of recolonisation of meio- and macrofaunal communities similar?

Materials and methods

Study area

The study area is the Titchmarsh Marina which is located within Hamford Water, Essex, UK, which is part of the so-called Walton Backwaters (figure 4).

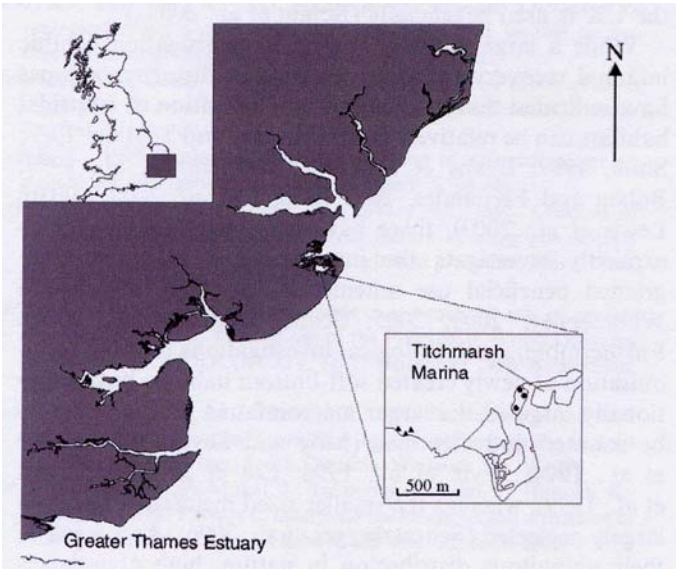

Figure 4. Location of the Titchmarsh Marina and the positions of the reference (triangles) and recharge (circles) stations.

The inlet is bordered by sand and shingle spits which provide protection against wave action from the open sea. These spits have been eroding and migrating landwards across saltmarsh and, consequently, protection and creation of intertidal mudflats and saltmarshes are primary aims within this area. The marina experiences significant tidal deposition of fine sediment and this requires frequent maintenance dredging. In 2000, a licence was granted by the Department for Environment, Food and Rural Affairs (Defra) for the marina to place maintenance dredged material directly onto an adjacent area of eroded saltmarsh.

[11] Source: Bolam *et al.* (2006)

During May and June 2001, the material was dredged and deposited on the recharge area (60-80 cm vertical overburden). The recharge (and reference) area is predominantly sheltered from wave action and experiences very little tidal current. The recharged material comprised 92% water, while the particulate component comprised 95% silt/clay and 1.2% organic carbon (Annex 2). The deposited sediments were classed as 'uncontaminated' in terms of the concentrations of a range of trace metals and TBT (Annex 2).

Sampling and analyses
Three stations were positioned within the recharge area (figure 4) and three at a reference site (figure 4) within the same saltmarsh system, at equivalent tidal heights and away from the effects of the recharge. As the resulting tidal height of the scheme was below the limits of saltmarsh plant colonisation, high-level mudflat was the most appropriate reference to assess natural temporal variability in macro- and meiofaunal communities.

The macrofaunal and meiofaunal communities of the recharge and reference stations were sampled 3, 6, 9 (meiofauna only), 12, 18, 24, 30, 36 and 42 months after recharge using a 0.01 m^2 (15 cm depth; three replicates per station) and 0.002 m^2 (5 cm depth, one replicate per station) corer for macrofauna and meiofauna respectively. On return to the laboratory, macrofaunal and meifaunal species were collected from the samples.

On each sampling occasion, a surface scrape of the top 3 cm of the sediment at each station was taken for sediment analyses. These surface scrape samples were analysed for water content, organic carbon content and particle size distribution. Redox potential profiles (1, 2 and 4 cm; three replicates) were measured (mV) at each site.

Data analysis
Statistical techniques were used to show spatial and temporal differences in environmental variables at the study sites. These variables included water content, mean particle diameter, silt/clay content, sorting coefficient, total organic carbon content and redox potential at 1 cm depth.

Total abundance, number of species and diversity were calculated for benthic invertebrate communities collected over the 42-month period. Tests for community differences were conducted both spatially (between recharge and reference areas) and temporally (between sampling intervals) using statistical techniques. The relationship between environmental variables and univariate community attributes (e.g. number of individuals, number of species, diversity) was investigated using correlation analyses. In order to assess whether the assemblage development at the reference and recharge sites followed similar trajectories, correlation analyses ware performed. This allowed the investigation of directional changes in community structure over time with a significant correlation indicating comparable temporal trends at reference and recharge sites. A similar procedure was carried out comparing the development of meiofaunal nematode and macrofaunal communities.

Results

Sediment data

Although the dredged material recharged had a very high water content (91.7% ± 0.29 SE, n = 3), the material rapidly dewatered area to 62.3% (±2.9 SE, n = 3) water content after 3 months, comparable to values recorded at the reference site. However, at this time (after 3 months), the granulometric properties of the sediments at the recharge site were significantly different, being better sorted and generally finer. In general, the sediments of the recharge area had lower organic and water contents relative to those of the reference sites: silt/clay content, mean phi, 1 cm redox and sorting coefficient were not influential properties distinguishing recharge and reference sediments.

No consistent temporal or spatial response in terms of differences in sediment properties is observed. At the reference site large within-station differences were observed.

Invertebrate data
A total of 76 nematode species (meiofauna) were collected at Titchmarsh Marina, 59 of which occurred at both the reference and the recharge areas. Eleven species were exclusively found at the reference area while six species were present at the recharge area only. Total nematode abundance, number of species and species diversity did not differ significantly between reference and recharge sites at any sampling time. Total nematode abundance and number of nematode species at both reference and recharge sites were

significantly higher within the first 18 months and nematode species diversity within the first 12 months compared to subsequent sampling occasions.

Nematode assemblages (community structure) from the recharge and reference areas were clearly different throughout the study period. Furthermore, in agreement with results from the univariate analyses, the composition of both reference and recharge communities collected within the first 12 months differed considerably from those collected thereafter. The temporal changes in nematode assemblage structure were greater at the reference sites (average dissimilarity = 47%) than at the recharge sites (average dissimilarity = 37%), i.e., the reference communities were temporally more variable. Results from statistical tests revealed significant spatial and temporal differences. There was no sign of convergence between recharge and reference communities, i.e., average dissimilarity (43%) remained constant over the 42-month study period.

A total of 59 macrofaunal taxa were collected at Titchmarsh Marina, 32 of which occurred at both the reference and the recharge sites. Twenty-two taxa were exclusively found at the reference sites while five were present at the recharge sites only. Total macrofaunal abundance, number of species and diversity were all significantly higher at the reference site compared to the recharge site. Station-replication reveals that there was a large degree of between-station variability, especially at the reference site, i.e., there were instances of significant differences between reference stations for each univariate parameter. Within-site spatial variability was much less marked within the recharge site. In contrast to the strong temporal changes observed for the meiofauna, there were no significant temporal changes in the macrofaunal indices.

Clear differences between the macrofaunal communities at the recharge and reference areas are observed, as was the case for the univariate indices (total macrofaunal abundance, number of species and diversity). There were also strong temporal changes revealed in the ordination, particularly for the recharge communities in the first 24 months post-recharge. There were signs of temporal convergence between recharge and reference communities: average dissimilarity decreasing from 64% to 42% over the 42 months. A comparison of the two macrofaunal ordinations reveals that there was a large amount of spatial variability in community structure of the reference stations. While reference stations 2 and 3 were consistently different from the recharge communities, reference station 1 showed a much greater similarity.

The temporal trajectories in macrofaunal and meiofaunal communities (combined for recharge and reference sites) were significantly related, i.e., both macro- and meiofaunal communities exhibited similar temporal changes. There were also significantly similar patterns between recharge and reference sites for both meiofauna and macrofauna. The temporal trajectories of the meio- and macrofaunal communities at the recharge site were significantly related; however, the relationship was not significant for the reference site.

Mean particle diameter and silt–clay and sand content in the sediment had statistically significant relationships with univariate nematode community attributes. Finer sediments with high silt–clay contents were generally characterised by diverse nematode assemblages with high numbers of nematode individuals and species. Nematode community structure was best explained by a combination of mean particle size, sorting coefficient and organic carbon content

Macrofaunal indices showed significant relationships with a wider range of sediment properties. Lower silt/clay and water contents and mean phi, and increased sorting coefficient and carbon content led to increased macrofaunal abundances. Lower silt/clay contents and increased mean phi, sorting coefficient and carbon content increased macrofaunal species number. Macrofaunal community structure had relatively stronger relationships with sediment properties, being best explained by organic carbon content in the sediment.

Discussion

Although the material recharged at Titchmarsh Marina was very fluid-like (92% water), the sediments rapidly dewatered and rich and diverse macro- and meiofaunal assemblages were present within 3 months.

Although a small number of meiofaunal individuals were present, no macrofaunal individuals were observed in the material recharged so direct transfer of individuals is unlikely to be an important initial recolonisation mechanism. Furthermore, the depth of material recharged (60-80 cm) at Titchmarsh Marina far exceeded the overburden through which benthic invertebrates may vertically migrate. Therefore, both macro- and meiofaunal recolonisation occurred primarily through settlement (active and/or passive) of planktonic larvae (for some macrofaunal taxa) or postlarval juveniles. Certainly, for meiofauna, settlement following passive re-suspension has been reported to be the primary recolonisation process at other beneficial use schemes. Temporal changes then occur via continued settlement and reproductive activity of settled individuals.

There was rapid initial recolonisation by both macro and meiofauna in the early stages post-recharge and they showed signs of similar temporal trajectories at the recharge site. However, their relationships between recharge and reference sites contrasted. For example, the univariate attributes (abundance, number of species, diversity) of the meiofaunal communities showed very little signs of departure from those of the reference site, yet displayed significant temporal change, especially after 12–18 months. This temporal change was observed in both the recharge and reference meiofaunal communities, indicating that it reflected natural temporal variability rather than successional dynamics. Macrofaunal univariate indices (e.g. abundance, number of species, diversity), however, were relatively constant through time, yet were continually reduced relative to those of the reference site.

The multivariate community structures (both meiofauna and macrofauna) of the recharge site showed no tendency to converge with those of the reference site and temporal shifts in recharge site communities were similarly observed in those at the reference site. For both macro- and meiofauna, the temporal trajectory in the recharge site was similar to that in the reference site and, overall, macrofauna showed similar trajectories to meiofaunal changes. Therefore, natural influences were primarily responsible for explaining temporal changes in the recolonising communities (rather than effects due to impacts of sediment recharge).

Conclusions

The recharge of dredged material onto intertidal habitats during a beneficial use scheme represents a major disturbance and, initially, the area is more-or-less devoid of benthic invertebrates and comprises sediments with very different physico-chemical characteristics from those of surrounding areas.

The univariate indices of community structure (abundance, number of species, diversity) indicated that the meiofaunal community at the recharge site was never significantly different from that of a nearby reference area. This was not the case for the community structure of the macrofauna. The univariate indices (abundance, number of species, diversity) of the macrofauna of the recharge site were continually significantly below those of the reference area (although this was not the case for all reference stations).

Multivariate analyses revealed that macro- and meiofaunal community structures were always significantly different from those of the reference communities. However, Bolam *et al.* (2006) postulate that the lack of convergence of recharge towards reference communities for both macro- and meiofauana at the Titchmarsh Marine recharge site does not suggest that the communities are adversely affected but, rather that there were spatial differences in one or more influential physical properties (that might have resulted in the observed differences in recharge and reference sites). Therefore no attempt has been made to assess the extent to which the recolonising communities can be described as recovered.

In the light of their results, Bolam *et al.* (2006) advocate that (future) studies aiming to investigate the community development of beneficial use schemes should adopt large numbers of stations sampled singly, rather than smaller numbers of stations each samples several times (more spatial data than temporal data). In addition, Bolam *et al.* (2006) propose that assessing recovery of a beneficial use scheme should be undertaken using pre-defined criteria in addition to comparisons with a reference site.

Annex 5.4 Charleston Ocean Dredged Material Disposal Site in South Carolina, USA[12]

Introduction

The Charleston, South Carolina, Ocean Dredged Material Disposal Site (ODMDS) was designated by the United States Environmental Protection Agency in 1987. The site is used actively by the US Army Corps of Engineers to receive bottom sediments dredged from channel maintenance and deepening projects in the Charleston Harbor estuary.
Since 1987, authorized disposal activities have taken place within a large area that encompasses approximately 5.3×2.3 nautical miles. A 2.8×1.1 nautical mile site within the larger area was used for most disposal activity until it was discovered that dumping operations were affecting reef habitats within the western quarter of the area. Subsequently, all disposal was limited to a foursquare mile area located in the offshore portion of the larger area (figure 5).

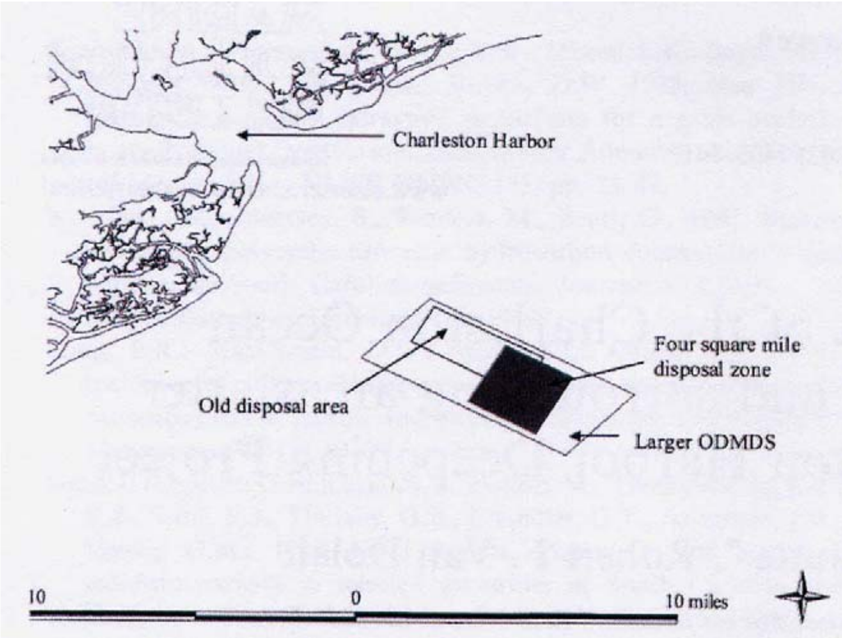

Figure 5. Location of the larger ODMDS, old disposal area, and current four square mile disposal zone (Source: Zimmerman *et al.*, 2003).

The US Congress authorized the most recent Charleston Harbor Deepening Project in 1996, with the project initiated in July 1999 and completed in April 2002. Approximately 20-25 million cubic yards of sediments were dredged and planned for disposal in the four square mile disposal zone (figure 5).

The US Army Corps of Engineers and the Marine Resources Research Institute studied the biological and physical conditions in the disposal site and surrounding areas:

1) prior to the deepening project (1993-1994);
2) midway thorough the deepening project (2000) and
3) following the completion of the deepening project (2002).

Data collected midway through the completion of the deepening project are summarized and compared to the baseline findings.

[12] Sources: Zimmerman *et al.*, 2003

Materials and methods

General conditions at dumping sites

Natural sediments in the area of the disposal site are fine-grained sands. The average water depth is 13 m. The currents in the disposal area are tidal, wind-driven and density-driven. Currents flowing toward the southeast or west could potentially transport dredged material toward the southwest corner of the ODMDS. Historical disposal in the area had resulted in the accumulation of fine-grained sediments west of the disposal zone.

Sampling

Sampling efforts took place within the permitted disposal zone and two zones (inner and outer) surrounding the disposal zone (figure 6). These three zones are composed of a total of 20 discrete strata of comparable size, approximately 1 square mile. Sampling was completed in September 2000. Within each of the 20 strata 10 randomly selected sites were sampled. Each sample was sub-sampled for analysis of sediment characteristics, for the presence of contaminants and for benthic species.

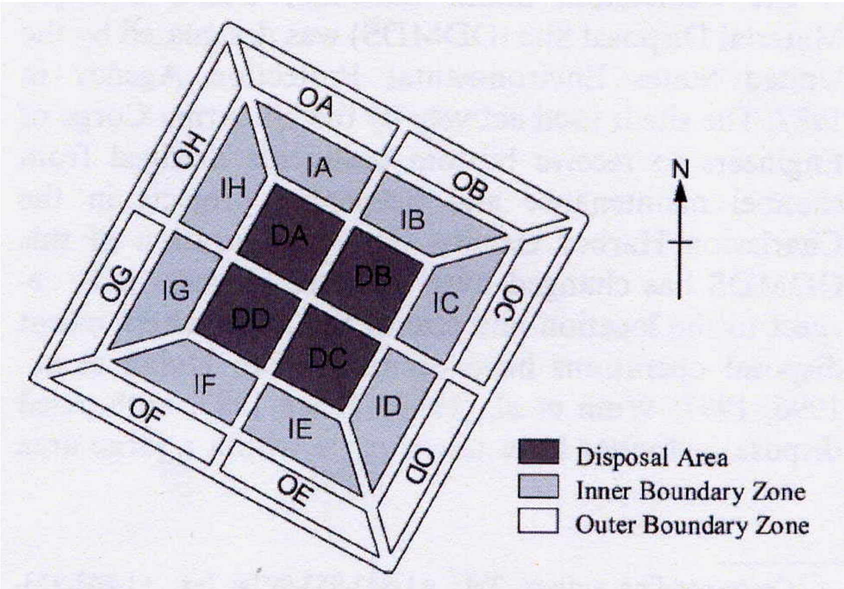

Figure 6. Permitted four square mile disposal zone and the surrounding boundary zones. Each strata is indicated with its two-letter code (Source: Zimmerman *et al.*, 2003).

Measurements

Sediment composition, mean grain size, and total organic carbon were analyzed in all samples (n=200). The sediment compositions were analyzed for percentages of sand, silt, clay, and calcium carbonate.
Composite sediment samples from each stratum were analyzed for metals, polycyclic aromatic hydrocarbons, polychlorinated biphenyls, and pesticides.

Benthic infaunal sorting and taxonomic identification were completed using a tiered approach. Samples were processed from a selected subset of strata collected in boundary areas known to be impacted based on findings from other studies conducted as part of an interim assessment. These strata (IA, IG, IH, OA, OG, OH) were then compared with samples from another subset of strata collected from the boundary zones where there was no evidence of change in sediment condition (IC, ID, OC, OD). Benthic organisms were sorted from each sediment sample, identified to lowest possible taxonomic level, and enumerated. Macrobenthic fauna collected within the disposal zone were not analyzed since the area greatly modified and primary management concerns were focused on possible impacts in the area surrounding the disposal zone.

Statistical analyses were performed to determine the impact of disposal of dredged sediment. For further details regarding the data analysis see Zimmerman *et al.* (2003).

Results

Although some contaminants were detected within the disposal area, none of the contaminant levels were above the effects range low levels (ERL). Because contaminant levels were not deemed biologically significant, they were not further discussed by Zimmerman *et al.* (2003).

Spatial comparisons

Disposal zone sediments had significantly more silt/clay than inner or outer boundary zones ($p<0.001$) due to the placement of fine-grained inner-harbor material in the disposal area. Silt/clay content in strata OG, DD, DB, OB and OH (figure 6) was greater than other strata (figure 6: OD, IB, IC, IA, OA). As a result of the disposal of fine-grained material, there was significantly less sand in disposal zone sediments than in inner or outer boundary zone sediments ($p<0.001$).

One portion of the disposal zone (figure 6; DA) and several strata on the western side (figure 6: OG, IG, OH, IH) had significantly smaller average grain size of the sand fraction than strata IE, OE, and OC (figure 6), which are located on the eastern and southern sides of the disposal zone.

Disposal zone sediments had significantly more organic matter than both the inner and outer boundary zones. In general, sediments on the western side and within the disposal zone (figure 6: i.e. DA, DD, OH, IH, OG, IG) had higher organic matter content than those on the eastern side (figure 6: IC, OC, ID, OD).

More than 15,700 organisms, of 402 taxa, were collected from the ten strata analyzed in 2000 that surrounded the disposal site. Statistical analyses of the benthic infaunal data revealed that the mean density and mean number of benthic (infaunal) species in areas west of the disposal zone were significantly lower than in the eastern strata ($p<0.05$). Cluster analysis of the data was conducted to evaluate the spatial differences in the benthic community inhabiting areas surrounding the disposal zone. The density and composition of the benthos in strata OG, IG, IH and OH (figure 6: impacted areas) were most similar to one another. Strata in the boundary areas to the east of the disposal zone in IC, OC, ID, and OD (figure 6: non-impacted areas) formed a second distinct cluster. The third cluster was composed of strata IA and OA, which are located to the northwest of the disposal area.

Temporal comparisons

Mean density and number of benthic (infaunal) species was lower in 2000 than in 1993 or 1994 in the majority of the impacted strata (figure 6: IA, IG, IH, OA, OG, OH), while there were no significant differences between mean density and number of species in 2000 and the earlier years in the majority of the non-impacted strata (figure 6: IC, ID, OC, OD).

The general taxonomic composition in the boundary zones also exhibited differences between 2000 data and the baseline data (1993/1994). Densities of the most abundant taxonomic groups, the polychaetes, showed no discernible pattern related to disposal activities. The density of molluscs and amphipods was significantly lower in 2000 than in 1993 or 1994 in the majority of the impacted and non-impacted strata. Organisms in the ‘other taxa’ category had significant lower densities in 2000 than in 1993 or 1994 in the majority of the impacted strata, whereas none of the non-impacted strata had significant differences among years.

Discussion

Spatial impacts

The elevated silt/clay content in the disposal zone is the result of the disposal of inner-harbor sediment. The elevated silt/clay content in strata outside the disposal zone, is likely a result of the movement of fines from the disposal zone, in addition to the material placed here as a result of unauthorized dumps. Tracers studies based on gamma activity measurements and aluminum levels (as indicator of clay content in surficial sediments) confirm that disposed sediments are dispersed to the areas west and northwest of the actual disposal site.

The benthic community analyses portion of the study of Zimmerman *et al.* (2003) was designed to determine the biological impacts of the dispersion of fines. Within the 2000 data set, the benthic community west of the disposal zone (approximately 1.5 mile from the disposal zone) was negatively impacted by disposal. Both density and number of species were significantly lower in impacted areas (figure 6: IA, IG, IH, OA, OG, OH) compared to areas not impacted by disposal (figure 6: IC, ID, OC, OD).

Temporal impacts

Temporal comparisons of the 2000 benthic community to the baseline benthic community (1993/1994) revealed that total density and number of species were affected by disposal. Those boundary areas into which dredged materials were dispersed, experienced a decline in total density as well as the number of species while boundary areas not affected by dredge materials showed no difference in the parameters between the present study (2000) and baseline values (1993/1994).Direct burial will often result in the immediate mortality of benthos. Habitat alterations can (e.g. change in sediment characteristics), however, have more long-term effects on the benthic community. In the area studied by Zimmerman *et al.* (2003) habitat alterations occurred due to the disposal of fine grained sediments on medium-coarse sand. This can explain the low abundance of the cephalochordate *Branchiostoma* sp in the impacted strata. This organism is known to prefer sandy sediments and is seldom found in muddy areas, such as the impacted area (west of the disposal zone).

Change in habitat also affected P. *dayi*. This organism had greater densities in the impacted areas in 2000 than in 1993/1994. P. *dayi* is a spionid polychaete classified as both a deposit feeder and a suspension feeder, depending on the availability of suspended and deposited particles. Larger abundances of this organism in the impacted area might be the results of the increased organic matter content in the impacted area.

Conclusions

The disposal of fine-grained inner harbour sediments into the Charleston ocean disposal zone (USA) has resulted in physical and biological effects in areas surrounding the disposal zone. Silt/clay content and organic matter content to the west of the disposal zone (approximately 1.5 mile from the disposal zone) were elevated above typical levels in near shore South Carolina waters. Changes in sediment characteristics have, in turn, impacted the disposal zone. These alterations in the benthic community were attributed to changes in bottom habitat characteristics rather than pollution effects.

Annex 6 Management options for disposal of dredged sediment (Source: adjusted after Essink, 1999)

Options for disposal	Possible effect	Cause
Near eelgrass beds	Deterioration of eelgrass	Increased turbidity
	Possible deterioration of eelgrass	Enhanced sediment deposition
Near intertidal flats	Decreased production by microphytobenthos depending on sedimentation on intertidal flats. Mortality among meiofauna depending on extent and type of sedimentation on intertidal flats. Mortality among macrobenthos depending on extent and sedimentation type on intertidal flats.	Enhanced sediment deposition
In tidal channels and other subtidal areas	Mortality among meiofauna depending on extent and type of sedimentation on intertidal flats. Mortality among macrobenthos depending on extent and sedimentation type on intertidal flats.	Enhanced sediment deposition
Outer estuary	Some damage to visual predators, fish (e.g. herring, sprat, smelt)	Increased turbidity
Near mouth of estuary	Reduced foraging for visual predators, birds (e.g. sandwich tern)	Increased turbidity
	Possible negative effect on zooplankton. Negative effect on growth of filter feeding benthos if increase of SPM > 20%.	Increased SPM concentrations
In upper estuary or near tidal watershed	Some decrease in phytoplankton production	Increased turbidity
	No negative effect on zooplankton (?). Less negative effect on filter feeding benthos.	Increased SPM concentrations
In spring/summer	Decrease effect on phytoplankton production. Negative effect on visual predators.	Increased turbidity
	Macrozoobenthos susceptibility larger than in winter	Enhanced sediment deposition
In winter	No decrease of phytoplankton production. Negative effect on herring in upper estuary.	Increased turbidity
	Little chance for macrofauna to escape from burial. Bad for winter migration of species like juvenile *Macoma balthica.*	Enhanced sediment deposition

In autumn	No decrease of phytoplankton production. No negative effect on visual predators.	Increased turbidity
	Chance for macrofauna to escape from burial. Normal winter migration of species like juvenile *Macoma balthica.*	Enhanced sediment deposition
In autumn/winter	No negative effect on filter-feeding benthos	Increased SPM concentrations
Use site each year	Incomplete recovery in-between dumping occasions	Enhanced sediment deposition
Each year different site	Better recovery of benthos at disposal sites	Enhanced sediment deposition